-DRAFT-

SCREENING ECOLOGICAL RISK ASSESSMENT FOR RICHARDSON FLAT TAILINGS PARK CITY, SUMMIT COUNTY, UTAH

February 2002

Prepared for the:

United States Environmental Protection Agency
Region VIII
999 18th Street, Suite 500
Denver, CO 80202

Prepared by:

Syracuse Research Corporation
Environmental Science Center - Denver
999 18th Street, Suite 1975
Denver, CO 80202

-DRAFT-

SCREENING ECOLOGICAL RISK ASSESSMENT FOR RICHARDSON FLAT TAILINGS PARK CITY, SUMMIT COUNTY, UTAH

February 2002

Prepared for the:

United States Environmental Protection Agency
Region VIII
999 18th Street, Suite 500
Denver, CO 80202

Prepared by:

Syracuse Research Corporation
Environmental Science Center - Denver
999 18th Street, Suite 1975
Denver, CO 80202

This page intentionally left blank.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF FIGURES
(Continued)

LIST OF TABLES

LIST OF TABLES
(Continued)

LIST OF APPENDICES

LIST OF ACRONYMS AND ABBREVIATIONS

ASARCO	American Smelting and Refining Company
ATSDR	Agency for Toxic Substances and Disease Registry
AUF	Area Use Factor
AVS	Acid Volatile Sulfide
AWQC	Ambient Water Quality Criteria
BAF	Bioaccumulation Factors
BCC	Bioaccumulative Contaminant of Concern
BG	Background
BJC	Bechtel Jacobs Company
BOM	Bureau of Mines
BSAF	Biota-Sediment Accumulation Factors
BW	Body Weight
CCME	Canadian Council of Ministries of the Environment
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act Information System
COPC	Contaminant of Potential Concern
DF	Dietary Fraction
dw	Dry Weight
E&E	Ecology & Environment, Inc.
EC50	Effective Concentration for 50% of the Study Organisms
ED50	Effective Dose for 50% of the Study Organisms
EPA	United States Environmental Protection Agency
EPC	Exposure Point Concentration
ERA	Ecological Risk Assessment
ERAGS	Ecological Risk Assessment Guidance for Superfund
ERL	Effects Range Low
ERM	Effects Range Median
GLWQG	Great Lakes Water Quality Guidance
GW	Groundwater
HI	Hazard Index
HQ	Hazard Quotient
HRS	Hazard Ranking System
IR	Ingestion Rate
IRIS	Integrated Risk Information System
LC50	Lethal Concentration for 50% of the Study Organisms
LOAEL	Lowest Observed Adverse Effect Level
LOEC	Lowest Observed Effect Concentration
MW	Monitoring Well
NEC	No Effect Concentration
NOAA	National Oceanic and Atmospheric Administration
NOAEL	No Observed Adverse Effect Level

LIST OF ACRONYMS AND ABBREVIATIONS
(Continued)

NPL	National Priorities List
OEA	OEA Research, Inc
ORNL	Oak Ridge National Laboratory
PCV	Park City Ventures
PEC	Probable Effects Concentration
PEL	Probable Effects Level
RCRA	Resource Conservation and Recovery Act
RF	Richardson Flat
RFD	Reference Dose
RFT	Richardson Flat Tailings
RI/FS	Remedial Investigation/Feasibility Study
RMC	Resource Management Consultants
SAP	Sampling and Analysis Plan
SCM	Site Conceptual Model
SEC	Sediment Effects Concentration
SEM	Simultaneously Extractable Metals
SERA	Screening Ecological Risk Assessment
SET	Severe Effects Threshold
SQG	Sediment Quality Guidelines
SW	Surface Water
TAL	Target Analyte List
TDS	Total Dissolved Solids
TEC	Threshold Effect Concentration
TEL	Threshold Effects Level
TMDL	Total Maximum Daily Load
TRV	Toxicity Reference Value
TSS	Total Suspended Solids
UCL	Upper Confidence Limit
UF	Uncertainty Factor
UPCM	United Park City Mines
URS	URS Operating Services, Inc.
USC	Upper Silver Creek
USDOI	United States Department of the Interior
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
XRF	X-Ray Fluorescence

1.0 INTRODUCTION

related

1.1 Purpose

This document is a screening level evaluation of potential risks to ecological receptors at the Richardson Flat Tailings (RFT) Site located near Park City, Utah (Figure 1-1). The purpose of the Screening Ecological Risk Assessment (SERA) is to identify the potential for adverse effects (risks) to ecological receptors resulting from exposure to contaminants released as a result of past mining activities. If potential risks are identified, then a more detailed Baseline Ecological Risk Assessment (ERA) may be warranted. The SERA process consists of four general steps: Problem Formulation, Exposure Assessment, Effects Assessment, and Risk Characterization (Figure 1-2).

The screening level problem formulation and risk characterization results are used to identify: 1) the need for a more detailed assessment; and, 2) the specific types of data needed to complete a more detailed assessment. The SERA is not intended to support any final quantitative conclusions about the magnitude of potential ecological risks identified in the screening-risk procedure(s).

1.2 Scope

This SERA is completed in accordance with current United States Environmental Protection Agency (USEPA) guidance for performing ecological risk assessments, in general (USEPA, 1998 and USEPA, 1992), and specifically, under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) (USEPA, 1997). The SERA is completed according to the recommended eight-step process presented in the Ecological Risk Assessment Guidance for Superfund (ERAGS) (Figure 1-3). Figure 1-3 is shaded to show which portions of the ERAGS process are addressed by this document for the RFT Site.

In accordance with USEPA guidance, this SERA is intentionally simplified and conservative. The conservatism allows for elimination of only those contaminants, receptor pathways and environmental media that are below a level of concern and for which there is high confidence of no adverse effects (risks). However, if the SERA indicates that contaminant concentrations in a particular medium are within a range of concern, it is appropriate to conclude that a potential for risk does exist and that a more refined ecological risk evaluation is needed to identify and quantify the actual risk(s).

1.3 Organization

The SERA is organized into ten sections. In addition to this introductory section, the SERA contains the following chapters or sections:

Section 2 This section provides the site characterization, which includes the site location, description, regulatory history, and environmental setting.

Section 3 This section provides a description of the available analytical data for the RFT Site including the nature and extent of contamination present in tailings, soils , surface water, sediments, and seeps (groundwater).

Section 4 This section provides the screening level problem formulation which includes discussions about the site conceptual model (SCM) selection of contaminants of potential concern (COPCs), and identification of assessment and measurement endpoints.

Section 5 This section presents the screening level ecological exposure assessment for aquatic invertebrates, fish, amphibians, terrestrial plants, soil invertebrates, and wildlife receptors.

Section 6 This section presents the screening level ecological effects assessment for aquatic invertebrates, fish, amphibians, terrestrial plants, soil invertebrates, and wildlife receptors. This includes descriptions of toxicity screening benchmarks for aquatic receptors (invertebrates, fish and amphibians) for surface water, seeps and sediments and for terrestrial plants and soil invertebrates for soils. The ecological effects assessment for wildlife identifies toxicity reference values (TRVs) or doses of contaminants by ingestion that are associated with no observed adverse effects or a lowest observed adverse effect.

Section 7 This section presents the screening level risk characterization for aquatic invertebrates, fish, amphibians, terrestrial plants, soil invertebrates, and wildlife receptors. aquatic and terrestrial wildlife receptors.

Section 8 This section presents and discusses the uncertainties associated with each of the steps of the SERA.

Section 9 This section discusses the data gaps present in the SERA and provides recommendations for the collection of data and analyses for completing a more detailed or baseline ecological risk assessment (ERA). The recommendations are based on the findings of the SERA.

Section 10 This section presents references used in the SERA.

2.0 SITE CHARACTERIZATION

2.1 Site Location

The RFT Site is located 1.5 miles northeast of Park City, Utah occupying about 700 acres in a small valley in Summit County, Utah (Figure 1-1). The RFT site is part of the Park City Mining District where silver-laden ore was mined and milled from the Keetley Ontario Mine as well as other mining operations (RMC, 2001a). Tailings were deposited into an impoundment covering 160 acres of the 700 acre property just east of Silver Creek. Tailings were deposited to the impoundment from the mill by use of a slurry pipeline from 1975 through 1981. Mining and milling operations ended in 1982.

2.2 Site Description

Tailings were first placed on RFT Site prior to 1950 (RMC, 2000a). Historical aerial photos confirm that tailings have been present at the flood plain tailings pile as early as 1953 (USEPA, 1991). The mill tailings present consist of mostly of sand-sized particles of carbonate rock with some minerals containing silver, lead, zinc and other metals. Few specific details are available concerning the configuration and operation of the historic tailings pond (prior to 1950) but certain elements are apparent. From time to time, tailings were transported to the Site through three distinct low areas on the southeast portion of the Site. Over the course of time, tailings materials settled out into the low areas that were ultimately left outside and south of the present impoundment area constructed in 1973 to 1974 (RMC, 2001b).

In 1970, Park City Ventures (PCV), a joint venture partnership between Anaconda Copper Company and American Smelting and Refining Company (ASARCO) entered into a lease agreement with United Park to use the Site for disposal of additional mill tailings from renewed mining in the area. PCV contracted with Dames & Moore to provide construction specifications for reconstruction of the Site for continued use as a tailings impoundment (Dames & Moore, 1974). The state of Utah approved the Dames & Moore plan and the current impoundment area was constructed in 1974 (RMC, 2000a). Before disposing of tailings on the Site, PCV installed a large earthen embankment along the western edge of the existing tailings impoundment and constructed perimeter containment dike structures along the southern and eastern borders of the impoundment to allow storage of additional tailings. PCV also installed a diversion ditch system along the higher slopes north of the impoundment and outside of the containment dike along the east and south perimeter of the impoundment to prevent surface runoff from surrounding land from entering the impoundment (RMC, 2001b). Dames & Moore recommended that special engineered seepage control devices be installed at the base of the main embankment. PCV did not follow this recommendation (Dames & Moore, 1974).

PCV conveyed tailings to the impoundment by a slurry pipeline from its mill facility located south of the Site. Over the course of operation, approximately 420,000 tons of tailings were disposed of at the Site. PCV failed to follow recommendations for disposal of the slurry in the impoundment (to place tailings along the perimeter of the impoundment and move towards the center) and placed a large volume of tailings near the center of the impoundment in a large, high-profile, cone-shaped feature. After cessation of operations in 1982, the presence of the cone-shaped feature resulted in prevailing winds form cutting into the tailings and the tailings becoming wind-borne (RMC, 2001b).

The RFT Site is currently under the ownership of United Park City Mines (UPCM) (RMC, 2000a). UPCM is a consolidation of Silver King Coalition Mines Company and Park Utah Consolidated Mines Company, formed in 1953 (RMC, 2000a).

2.2.1 Sources

Do not include Flood Plain Tails

There are two known sources of contamination at the RFT Site. These include the tailings impoundment previously described and a flood plain tailings pile. The flood plains tailings pile is located immediately west of the tailings impoundment and covers about 6 acres along the banks of Silver Creek (USEPA, 1991). This source is reported to be located on the western side of Silver Creek about 300 feet upstream of the confluence of Silver Creek with the wetland area and extends from there for about 2500 feet upstream. The USEPA and the State of Utah have both observed tailings entering Silver Creek from the flood plain tailings pile (USEPA, 1991). According to analyses performed in 1985 and 1989, the flood plain tailings pile contains arsenic, cadmium, copper, lead, mercury, silver, and zinc (USEPA, 1991).

2.2.2 Site Features

The Focused Remedial Investigation/Feasibility Study (RI/FS) Workplan prepared by RMC in May 2000, provides detailed information on the RFT Site features (Figure 2-1). Information pertaining to the main embankment and containment dikes, the diversion ditches and off-impoundment tailings is summarized in the following subsections.

2.2.2.1 Main Embankment and Containment Dikes

The majority of the tailings at the RFT Site are contained in a closed basin, with a large, earth, embankment in place along the western edge of the Site (Figure 2-1). The "main embankment" is vegetated and is approximately 40 feet wide at the top, 800 feet long, and has a maximum height of 25 feet. This embankment is designed to allow water to seep from the impoundment to relieve hydraulic pressure on the embankment. Currently, surface water is present in the form of a seep located near the north end of the base. A series of man-made containment dikes contain the tailings along the southern and eastern perimeter of the impoundment. The northern edge of the impoundment is naturally higher than the perimeter dikes (RMC, 2000a).

2.2.2.2 Diversion Ditches

A diversion ditch system borders the north, south, and east sides of the impoundment to prevent runoff from the surrounding land from entering the impoundment. Precipitation falling on the impoundment area creates a limited volume of seasonal surface water (Figure 2-1). The north diversion ditch collects snowmelt and storm water runoff from upslope, undisturbed areas north of the impoundment and carries it in an easterly direction towards origin of the south diversion ditch. An unnamed ephemeral drainage to the southeast of the impoundment also enters the south diversion ditch at this point. Additional water from spring snowmelt and storm water runoff enters the south diversion ditch from other areas lying south of the impoundment at a point near the southeast corner of the diversion ditch structure. Water in the south diversion ditch flows from east to west and ultimately empties into Silver Creek just upstream of Highway 189 near the north border of the Site. Water flow from the south diversion ditch into Silver Creek occurs during the higher water periods of the year (RMC, 2000a). All year?

2.2.2.3 Off-Impoundment Tailings

Additional tailings materials are present outside and to the south of the current impoundment area. During historic operations of the tailings pond, tailings accumulated in three naturally low areas adjacent to the property that eventually became the impoundment. In the 1970s, when PCV constructed the perimeter dike and diversion ditch along the south perimeter of the impoundment, tailings present in the three low areas were left in place, outside of the present impoundment. Starting in 1983, United Park reportedly covered most of these tailings outside of the current impoundment with a low permeability, vegetated soil cover. Other types of clean fill material, imported from construction work in Park City, were also used to cover the tailings outside of the impoundment. The cover in some of these areas is reported to be as thick as 10 to 15 feet (RMC, 2000a). However, recent surveys of off-impoundment cover soils indicate that at some locations soil cover is absent leaving exposed surface tailings and in other places the soil cover is less than a few inches (RMC, 2001a).

2.2.3 Site Activities

UPCM and others have conducted certain efforts at the RFT Site to support investigation of integrity or closure. These activities are briefly described in the following subsections.

2.2.3.1 Impoundment Integrity Analyses

Noranda Mining, Inc. (Noranda) leased the RFT Property from UPCM in 1980 (RMC, 2000a). Shortly after Noranda entered into the lease agreement, Dames & Moore was contracted to conduct an impoundment integrity investigation. Although several construction flaws are noted, including the oversteeping of the main embankment along various locations, Dames & Moore concludes that the main embankment and containment dikes are in no immediate threat of failure. Dames & Moore once again recommends the installation of seepage control systems at the base of the main embankment (RMC, 2000a). Noranda does not follow this recommendation. Noranda disposed of 70,000 tons of additional tailings material and ceased operations in 1982. No new tailings have been placed at the Site since that time (RMC, 2000a).

2.2.3.2 Soil Cover of Tailings

Starting in 1983, UPCM began placing soil cover on tailings outside of the impoundment, located in three low areas south of the south diversion ditch (Figure 2-1). By 1985, the tailings impoundment had dried out enough in certain areas to support heavy equipment and UPCM began installing soil cover material over those portions. The cover soils are reported to be clay-rich and came from both the Park City area and from within the RFT Site (RMC, 2000a).

Between 1985 and 1988, UPCM also placed soil cover around the cone shaped tailings structure inside the impoundment area at locations where it had dried out enough to support heavy equipment. The primary objective of placing the soil cover was to prevent prevailing winds from cutting into the cone-shaped tailings By 1988, this work was completed and UPCM began a more aggressive program to cover all exposed tailings. It is reported that at least 12 inches of low-permeability, clay cover material was placed in the impoundment and that the soil cover was then vegetated (RMC, 2000a). More recent inspection of the cover soils at the main impoundment and off-impoundment indicate a shallow soil cover in some areas (less than 12 inches) and no soil cover in other locations (RMC, 2001a).

By 1992, to soil cover work was completed (RMC, 2000a). Shortly after completion, E&E (1993) completed a soil depth survey within the impoundment and an inspection of the main embankment. X-Ray Fluorescence (XRF) was used to confirm the visual contrast between top soil and the tailings below (E&E, 1993). E&E (1993) determined that on average, cover soils varied between less than 6 inches and 14 inches in depth. Areas in which cover soils were known to be more than 3 feet in depth were not surveyed. For the 29 locations studied, one exhibited exposed tailings. As a result, UPCM placed additional soil in this area (RMC, 2000a). More recent soil cover surveys for the main impoundment, however, indicate that at some locations the soil cover is less than 12 inches in depth (RMC, 2001a; 2001b).

2.2.2.3 Wedge Buttress Reinforcement

In an effort to correct the over-steepened portions of the main embankment, UPCM proposes to design the installation of a wedge buttress. The buttress will enhance the long-term effectiveness of the final closure remedy for the Site. UPCM will evaluate the condition of the main embankment during the RI/FS, and then prepare construction design specifications as part of the final remedial design process. Data from the seep located at the base of the main embankment may need to be gathered in order to develop an appropriate wedge buttress design (RMC, 2000a).

2.2.2.4 Fencing

In the mid 1980's, UPCM installed a fence along most of the Site boundary, including the entire impoundment and much of the property south of the impoundment. The fence was placed to restrict access to the Site. UPCM reports it will maintain the fence in good repair and will continue to control Site access until such time limited access is no longer necessary (RMC, 2000a).

2.2.2.5 Diversion Ditch Reconstruction

In 1992 and 1993, UPCM reconstructed the south diversion ditch by decreasing the slope of its banks from nearly vertical to a more gradual slope. UPCM placed a clay soil cover over the re-sloped banks down to and including areas of the banks underwater. The existing ditch banks were re-vegetated and the bottom of the ditch was not disturbed during these efforts. In May of 1999, United Park reconstructed the north diversion ditch along its entire length in the same manner (RMC, 2000a).

2.3 Regulatory History

The RFT Site was first proposed for the National Priorities List (NPL) on June 24, 1988. The original Hazard Ranking System (HRS) score of 50.23 was based on surface water and air migration pathways (USEPA, 1991). Areas evaluated in the HRS included the impoundment and adjacent areas (USEPA, 1991). Based on public comments, the site was dropped from consideration for the NPL on February 11, 1991 (USEPA, 1991). The HRS scoring criteria for surface water migration pathways were revised in 1992. The USEPA is currently proposing the site for a second NPL consideration under the revised HRS (USEPA, 1991). Along with the impoundment area and adjacent areas, the new proposal includes the Park City Municipal Landfill and the Silver Creek flood plain area (RMC, 2000a).

Define Site Boundary

2.4 Site Environmental Setting

2.4.1 Topography and Surrounding Land Use

The site is located in a rural area whose topography is characterized by a broad valley with undeveloped rangeland. Silver Creek is located within a few hundred feet from the main tailings impoundment. This perennial stream drains other historic tailing ponds in the Park City area (Mason, 1989). Silver Creek originates in an upper mountain zone where access is limited to recreational users. As Silver Creek passes through Park City and in the surrounding suburban areas, the land use is primarily residential and commercial changing to recreational and agricultural downstream to its confluence with the Weber River (RMC, 2001a).

2.4.2 Geology and Hydrogeology

2.4.2.1 Geology

The RFT Site is located in the Wasatch Range Section of the Middle Rocky Mountain Physiographic Province in north-central Utah in an area composed of a complex fold and thrust belt that is covered over with igneous rock (RMC, 2000a; 2000b). The sedimentary bedrock, which dates to the Paleozoic and Mesozoic age, is covered by a thick layer of extruded igneous rock that dips approximately 25 to 60 degrees to the north and strikes northeast-southwest (Bromfield and Crittenden, 1971). Tertiary gravels and igneous rocks cover the Mesozoic sedimentary rocks (RMC, 2001a). There are no known faults near the RFT Site.

Alluvial and colluvial sediments lie 30 to 50 feet deep beneath the tailings on site. These sediments are product of the erosion of neighboring and underlying igneous extrusions. Borehole data has shown that these sediments consist of: 2-5 feet of soft, organic, and clay rich topsoil; 1-30 feet of mixed fine-grained silt and clay; 4 feet of sand and gravel; highly weather, volcanic breccia which is composed of soft, tight, sandy and silty clay grading to harder fractured volcanic rock (RMC, 2000b). The unconsolidated valley fill is reported to range in thickness from a few feet adjacent to hills and mountains to at least 260 feet, centrally in valleys (Mason, 1989)

2.4.2.2 Hydrogeology

In 1999, UPCM contracted Weston Engineering, Inc. (Weston) to conduct a hydogeological survey of the site. The hydrogeology in the area consists of shallow alluvial aquifers located in the alluvial and colluvial material as well as the deeper Silver Creek Breccia bedrock aquifer located in the Keetley volcanics (RMC, 2000b). The shallow aquifers are found fifteen to thirty feet below ground surface in gravelly clay. The shallow aquifers' hydraulic gradients parallel topography (south to north) except at the southern boundary of the tailings embankment where flow changes to the northwest due to diversion ditches. The hydrogeology of the Site area has been described in a separate report (Weston, 1999).

2.4.2.3 Hydrology

Silver Creek flows approximately 500 feet from the main embankment along the west edge of the Site (RMC, 2000a). The headwaters of Silver Creek are comprised of three major drainages in the Upper Silver Creek Watershed; the Ontario Canyon, the Empire Canyon and Deer Valley. Flows from Ontario and Empire Canyons occur in the late spring to early summer months in response to snowmelt and rainfall, while Deer Valley flows appear to be perennial and originate from snowmelt and springs (RMC, 2000b). Surface water runoffs for this watershed are lower than that of comparable mountain watersheds which are less fractured and may have a more developed layer of unconsolidated materials (Brooks et al., 1998). Overall, runoff and precipitation flows from Empire and Ontario Canyons are low compared to the substantially large flow contributed by Deer Valley (USEPA, 2001a). The major influence on water flow in Silver Creek near the RFT Site is the Pace-Homer (Dority Springs) Ditch, which derives most of its flow from groundwater (USEPA, 2001a). The outflow from the Pace-Homer Ditch enters Silver Creek at several locations across the Prospector Square area. Significant riparian zones and wetlands exist near the RFT Site in areas that historically consisted of accumulated tailings piles. and below

2.4.3 Climate

High for RF

Richardson Flat is located in north-central Utah. The average monthly precipitation is approximately 3.64 inches with an average annual precipitation of 43.68 inches (www.weather.com - accessed 08/5/01). The average monthly temperature ranges from 19°F to 58°F. with an average for the year of 36°F. Elevations near the RFT Site range from 6,930 to 9,075 feet above sea level (RMC, 2000b).

2.4.4 Ecology

There is very limited information concerning the biological communities present at the RFT Site. This section summarizes the information from reports available for review at the time of the SERA.

2.4.4.1 Aquatic Community

In accordance with the State of Utah surface water code, the Weber River from the Stoddard diversion to its headwaters (including Silver Creek) is classified as a cold water fishery (3A) and is protected for cold water species of game fish and other cold water aquatic life, including the necessary aquatic organisms in the food chain. Elevated zinc concentrations, in comparison to the State aquatic life standard for 3A designated streams, have consistently been reported in Silver Creek.

According the public health assessment conducted by ATSDR there are few studies available concerning fish in Silver Creek. A survey conducted in 1954 found a small number of trout in Silver Creek (ATSDR, 1994) but in 1970, fish were not present during electroshocking (ATSDR, 1994). More recently, biologists have reported cutthroat troat in Silver Creek, however, information regarding number of individuals or sampling locations are not available (E&E, 1991). A 1986 investigation produced no fish but pan-sized trout were reportedly seen in Silver Creek near the RFT Site in the spring of 1992 (USEPA, 1993c; ATSDR, 1994).

Visual observation

2.4.4.2 Terrestrial Community

There was no information located pertaining to the plant and terrestrial wildlife communities (mammals and birds) present at the RFT Site.

2.4.4.3 Threatened or Endangered Species

Federally listed threatened or endangered wildlife species that are known or are suspected to inhabit Summit County include the bald eagle (*Haliaeetus leucocephalus*), the Canada lynx (*Lynx canadensis*) and possibly the whooping crane (*Grus americana*) and the black-footed ferret (*Mustela nigripes*) (Utah Division of Wildlife website - accessed 08/03/01). No threatened or endangered plant species were identified.

3.0 DATA SUMMARY AND EVALUATION

The SERA is based on the available analytical and physical data from investigations completed within the RFT Site area. A summary of the raw data is provided as Appendix A. These results represent the known nature and extent of contamination and are used as the basis of the SERA.

3.1 Tailings Data

As previously discussed, contamination at the RFT Site originated from the deposition of tailings within and outside of an impoundment. In July 1989, one tailings sample from the main impoundment area (stratified depths from 1-18 inches) and five tailings samples (0-6 inches) from flood plain areas were collected and data were presented in the HRS (USEPA, 1991). These samples were analyzed for total arsenic, cadmium, copper, lead, mercury, silver and zinc.

In May 2001, RMC collected tailings samples from the three locations within the impoundment at 1 foot depth intervals (beginning from the bottom of the cover soils to a depth of 5 feet). Figure 3-1 identifies these locations as green circles on the impoundment. Samples were analyzed for aluminum, antimony, arsenic, cadmium, chromium, copper, iron, lead, mercury, selenium, silver, and zinc. These samples were collected to evaluate the long-term fate of metals in tailings and the chemical stability of the tailings (RMC, 2001a).

Tailings disposal is also present in areas located outside the impoundment (Figure 3-1) but the spatial extent of these areas are not well defined. In June 2001, RMC collected tailings samples from locations south of the south diversion ditch in an effort to determine the extent of tailings disposal. This study was also completed to evaluate soil cover thickness, and if the tailings were contributing to zinc concentrations in the south diversion ditch. Samples were analyzed for aluminum, antimony, arsenic, cadmium, chromium, copper, iron, lead, mercury, selenium, silver, and zinc.

Analytical results for these three data sets are provided in Table 3-1. In order to evaluate the most current site conditions, the tailings data collected in July 1989 for the HRS are excluded from the SERA. Data included in the SERA are limited to tailings data collected by RMC through December 2001.

3.2 Soils Data

3.2.1 On-Impoundment Soils

In August 1992, Ecology & Environment, Inc. (E&E), under direction from EPA, investigated the RFT Site with respect to immediate threats to human health or the environment. The depth of soil cover was determined at 29 locations on the impoundment (based on an approximate grid pattern of 400 ft by 400 ft). At six of these locations, samples were analyzed for Target Analyte List (TAL) metals. These analytical results are presented in Table 3-2. Each of the samples, with the exception of sample RF-SO-3, are representative of cover soils on the impoundment in 1992. Sample RF-SO-3, was collected in an area of salt grass not yet covered by UPCM and is representative of tailings (E&E, 1993). Subsequently, UPCM placed additional soil cover in areas with thin cover (as identified by E&E, 1993) and on other areas to support site closure efforts (RMC, 2001a).

Currently, the cone-shaped tailings impoundment is reported to be covered with soil and vegetation with no areas of exposed tailings (RMC, 2001a). However, the extent, thickness, and chemical characteristics of the cover soils are not well defined. In May 2001, RMC collected 41 cover soils from 6 transects based on a 500 ft by 500 ft grid across the impoundment at a depth of 0-2 inches (distinct locations are identified as A through I). Figure 3-1 shows the locations at each grid node. Additional depth samples, ranging from 5 to 18 inches, were collected at 11 of these locations. All samples were analyzed for arsenic and lead with 20% of the samples analyzed for all RCRA metals. The analytical results for on impoundment cover soils are provided as Table 3-2.

In order to evaluate the most current Site conditions, the cover soils data collected by E&E in August 1992 are excluded from the SERA. The risk evaluation in the SERA is based on data for on-impoundment cover soils collected by RMC through December 2001.

3.2.2 Off-Impoundment Soils

Historically, prevailing winds from the southeast carried tailings from the impoundment and deposited them in the surrounding areas. In an effort to assess the extent and potential environmental impact of these wind-blown tailings, off-impoundment soil samples were collected from one transect north (T1) and two transects south (T2 and T3) of the RFT Site in May of 2001 (Figure 3-2). RMC collected eight distinct samples at T1 (A through H) and ten distinct samples at T2 and T3 (A through J) at two depth intervals (0-2 inches and 1-6 inches). All samples were analyzed for arsenic and lead with 20 % of the samples analyzed for all RCRA metals. Analytical results for these off-impoundment soils are provided as Table 3-3.

In September 2001, eight surface soil samples (0 to 2 inches in depth) were collected from locations surrounding the RFT Site to better determine the study area boundary (Figure 3-3). These samples were analyzed for arsenic and lead and the analytical results are provided in Table 3-3. Concentrations of arsenic and lead in sample SAB-6 are elevated compared to other results. Based on these results, it is assumed that this sample is representative of tailings and it is excluded from inclusion in the off-impoundment soils dataset (RMC, 2001b). The SERA is limited to off-impoundment soils collected by RMC through December 2001.

3.2.3 Background Soils

In order to determine the concentrations of metals in areas not affected by wind-blown tailings from the RFT Site, RMC collected background samples from areas not impacted by tailings deposition. It is important to note that these samples are representative of anthropogenic, non-site related levels, and do not represent "pristine" (not influenced by human activity) environmental levels.

Grab samples were collected at a depth of 0 to 2 inches from each of eleven locations (Figure 3-4) and were analyzed for arsenic and lead with 20% of the samples (BG8 and BG10) analyzed for all RCRA metals. The results are presented in Table 3-4. The arsenic and lead concentrations in sample BG11 are more than 30 times and 100 times greater, respectively, than those observed in other samples. This sampling location was later reported to be representative of tailings and is excluded from the background soils data set (personal communication, BTAG Mtg, 8/9/01).

Contact Letter implication?

3.3 Surface Water Data

Surface water data were compiled from five sources including E&E (1993), Utah water quality monitoring , USEPA (2001a), UPCM surface water monitoring, and RMC monthly sampling . A description of the surface water data from each source is provided in the following subsections.

For the purposes of conducting the SERA, surface water data from Silver Creek are segregated into two reaches; upstream and downstream of the RFT Site. To be consistent with the upstream/downstream designations used by UPCM, the cut-off point for these reaches is the rail trail bridge located northeast of State Highway 40 near the main embankment. In order to evaluate the most current site conditions, surface water data for the south diversion ditch are limited to samples collected after ditch reconstruction (1993 to present).

Ecology & Environment, Inc. (1993)

In August 1992, E&E collected surface water samples from Silver Creek and the south diversion ditch. As presented in Figure 3-5, six samples were collected along Silver Creek (RF-SW-1 to RF-SW-6) and two samples were collected from the south diversion ditch (RF-SW-7 and RF-SW-8). Analytical results for these surface water samples are provided as Table 3-5.

Utah Water Quality Monitoring (STORET)

Water quality monitoring data for several stations along Silver Creek were obtained electronically from an EPA STORET download query (Modernized Version). Data is available from nine locations on Silver Creek. Samples are collected and analyzed monthly for water quality parameters such as total hardness, pH, and temperature, as well as total recoverable and dissolved metals including arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc. Information for each of the Silver Creek stations is provided in the following text table. Analytical results are provided in Appendix A.

Station ID	Location Description	Latitude	Longitude	Sampling Dates
492674	Silver Creek at Farm Crossing in Atkinson	40.742167	-111.474167	12-Jan-68 to 13-Apr-00
492675	Silver Creek at Wanship above confluence with Weber River	40.813000	-111.401667	20-Dec-79 to 17-Jun-99
492676	Silver Creek 2 miles north of Atkinson	40.768500	-111.467667	21-Aug-81 to 11-May-89

Station ID	Location Description	Latitude	Longitude	Sampling Dates
492677	Silver Creek at I-80 Crossing at Atkinson east of Silver Creek Junction	40.743833	-111.473000	20-Dec-79 to 22-Jan-92
492679	Silver Creek at Waste Water Treatment Plant	40.735167	-111.474667	04-Jun-87 to 13-Jun-00
492680	Silver Creek above Atkinson	40.735167	-111.475167	17-Sep-81 to 13-Apr-00
492685	Silver Creek at US40 Crossing east of Park City	40.683000	-111.456000	02-May-75 to17-Jun-99
492694	Silver Creek at Railroad Crossing below Park City above Landfill	40.658000	-111.501833	20-Dec-79 to 28-Nov-83
492695	Silver Creek at City Park above Prospector Square	40.654333	-111.501667	06-Aug-97 to 17-Jun-99

USEPA (2001a) Silver Creek Watershed Sampling

In 2000, EPA completed an investigation of the Silver Creek watershed to better characterize the sources of heavy metals and to evaluate the total maximum daily load (TMDL) (Figure 3-6). A total of 31 surface water sampling locations are available from the watershed study for Silver Creek and its headwaters in Empire Canyon, Ontario Canyon, Deer Valley (Figure 3-7). For the purposes of the SERA only data from sampling stations on the lower reaches of Silver Creek (USC-1 through USC-7) below Prospector Square are used for the risk evaluation. Surface water samples for USC-4 were collected from the south diversion ditch on the RFT Site. Samples were collected in May and September 2000, respectively, to account for high (peak spring runoff) and low flow (fall or winter seasons). Some locations were re-sampled in November 2000 due to problems with mercury analysis. Average concentrations from each sampling location are provided in Table 3-6.

UPCM Monitoring

Since 1975, UPCM has collected surface water samples from the south diversion ditch (N5), and Silver Creek upstream (N4) and downstream (N6) of the confluence with the south diversion ditch (Figure 3-8). Surface water samples were collected monthly (usually from April to November) and analyzed for copper, cyanide, lead, mercury, manganese, zinc, total suspended solids (TSS) and total dissolved solids (TDS). The range of concentrations measured at each sampling location are provided in Table 3-7. At the time of the SERA, surface water data collected prior to April 1982 was not available for review.

RMC Monthly Sampling (RMC, 2001c)

Since May 1999, RMC collects monthly surface water from several locations along Silver Creek, the south diversion ditch, the unnamed drainages flowing into the south diversion ditch, and ponded areas at the RFT

Site. Specific locations are identified in Figure 3-9 and detailed station information is summarized in the following text table. Surface water samples were analyzed for total recoverable and dissolved TAL metals and water quality parameters. Average concentrations from each sampling location are provided in Table 3-8.

Station ID	Location Description	Sampling Dates
RF-1	Unnamed drainage flowing into the south diversion ditch	19-May-99 to 7-May-01
RF-2	South diversion ditch	19-May-99 to 7-May-01
RF-3	Unnamed drainage flowing into the south diversion ditch	19-May-99 only
RF-3-2	Unnamed drainage flowing into the south diversion ditch	4-Apr-01 to 5-Jun-01
RF-4	South diversion ditch	19-May-99 to 9-Jul-01
RF-5	South diversion ditch	19-May-99 to 7-Aug-01
RF-5-4	South diversion ditch*	4-Apr-01 to 7-May-01
RF-6	South diversion ditch	19-May-99 to 18-Sep-00
RF-6-2	South diversion ditch	9-Jun-99 to 3-Dec-01
RF-7	Silver Creek upstream of confluence with south diversion ditch	19-May-99 to 7-Nov-00
RF-7-2	Silver Creek upstream of confluence with south diversion ditch	9-Jun-99 to 3-Dec-01
RF-8	Silver Creek downstream of the confluence with south diversion ditch	19-May-99 to 3-Dec-01
RF-8-2	Silver Creek downstream of the confluence with south diversion ditch*	9-Jun-99 only
RF-9	Ponded water on the tailings impoundment	19-May-99 only
RF-10	Unnamed drainage flowing into south diversion ditch	9-Jun-99 only

*Assumed; actual sampling locations not provided on map.

3.4 Sediment Data

Sediment data are compiled for the SERA from three separate sources including E&E (1993), USEPA (2001a) and RMC monthly sampling. A description of the sediment data from each source is provided in the following text table.

Use of surface water data for the south diversion ditch in the SERA is limited to samples collected after ditch bank modification. This limitation is not, however, placed on the use of sediment data. During reconstruction, UPCM did not disturb the bottom of the ditch bed (RMC, 2001a) thus the existing sediments were not disturbed and constraining use of the is not necessary.

As with the surface water data set, Silver Creek sediments are designated as either upstream or downstream of the RFT Site using the same cut-off point for these reaches at the rail trail bridge located northeast of State Highway 40 near the main embankment.

Ecology & Environment, Inc. (1993)

In August 1992, E&E collected four sediment samples (RF-SD-01 to RF-SD-04) from the south diversion ditch "wetlands" area located at the base of the main embankment and Silver Creek (Figure 3-5). Water flow through this wetlands area is primarily from the south diversion ditch, although some seepage from the impoundment area may influence the flow and chemistry (E&E, 1993). Analytical results for these sediment samples are provided in Table 3-9. Based on the ratios of chemicals in tailings to compared to those in the wetlands sediments, E&E concluded that the sediments in the wetlands area are tailings material from the impoundment (E&E, 1993).

USEPA (2001a) Watershed Sampling

EPA collected sediment samples from 16 locations in the Silver Creek watershed (Figure 3-7). These samples were staggered across the watershed and co-located with specific surface water sampling sites to determine the relative level of metals throughout the system and evaluate interactions with surface water (USEPA, 2001a). At each location, both a surface and sub-surface (0-12 inches) sample was collected and analyzed for heavy metals. Data used in the SERA are limited to sampling stations on the lower reaches of Silver Creek (USC-1, USC-2, USC-5, USC-6, USC-7) below Prospector Square. Analytical results for these sediment samples are provided in Table 3-9.

RMC Monthly Sampling (RMC, 2001c)

In May 2001, RMC sampled sediments at six locations (RF-SD-1 to RF-SD-6) along the length of the south diversion ditch at a depth of 0 to 6 inches. Each sediment sample is designated by a blue 'X' in Figure 3-1. These samples were collected to evaluate the long-term effectiveness of the wetland system to remove metals in the water and to aid in the determination of the source of metals in water flowing from the diversion ditch (RMC, 2001a). Analytical results for the south diversion ditch sediments are provided in Table 3-9.

3.5 Seep Data

Because the main embankment is designed to allow water to seep from the impoundment to relieve hydraulic pressure, it is likely that metals leach from tailings into groundwater at the RFT Site. At the RFT Site, a small seep (flow of gallons per day) is located at the northern base of the main embankment (RMC, 2000a). Currently, no water or sediment data exist for this seep.

3.6 Groundwater Data

Since 1973, PCV and UPCM have collecting groundwater data quarterly from monitoring wells MW-1, MW-2, and MW-3 (RMC, 2000a). After their installation in 1976, PCV also began collecting groundwater from wells MW-4, MW-5, MW-6. E&E began collecting additional groundwater data in 1984 from a well (RT-1) installed up gradient of the main embankment. E&E also sampled the two existing down gradient monitoring wells MW-1 and either MW-5 or MW-6 . [It is unclear as to which well, MW-5 or MW-6, was sampled.] Well MW-2 was buried during the installation of wells MW-4, MW-5, MW-6 in 1976. The USEPA contracted E&E in 1992 to collect ground water samples from three additional locations (RF-GW-04, RF-GW-05, and RF-GW-09). The location of groundwater monitoring wells is provided on Figure 3-9.

Because measured seep concentrations are not available, measured concentrations from groundwater monitoring wells at the base of the main embankment near the seep are used to estimate seep water concentrations. Groundwater data is available for several site monitoring wells (MW-01, MW-03 through MW-06) located at the base of the main embankment. In addition, data from an upgradient monitoring well (RT-1) is used to estimate upgradient groundwater concentrations. The range of concentrations measured for these monitoring wells are presented in Table 3-10.

3.7 Biological Tissue Data

At the time of the SERA, the analyses of contaminant concentrations in biological tissues (aquatic or terrestrial) were not available from existing data reports and literature.

3.8 Summary of Analytical Data

Table 3-11 provides a summary of the analytical data available for the SERA. This table compares the analytical parameters available for the environmental media sampled and analyzed. As previously described, there are eight sources of sampling data including: RMC (2000a), EPA (1991); E&E (1993); EPA (2001a); RMC(2001a); RMC (2001c); UPCM and STORET. These programs do not have one common list of analytes for all environmental media. Table 3-11 provides a side-by-side comparison of the parameters available for each media type from each source of sampling data.

4.0 SCREENING LEVEL PROBLEM FORMULATION

Problem formulation is a systematic planning step that identifies the major factors to be considered in the SERA (USEPA, 1997). The problem formulation includes an evaluation of the fate and transport of contaminants of potential concern from waste sources to the receptors and identification of exposure pathways for the receptors. These factors are combined to present a site-conceptual model. Assessment endpoints are then defined and measurement endpoints developed that are the basis for the SERA. The site-conceptual model for the RFT Site was developed based on the ecological site conceptual model presented by RMC in the RI Sampling and Analysis Plan (RMC, 2001a). The revised ecological site conceptual model is described in the following subsections. Additions and changes made in comparison to the original model is discussed in Section 4.4.

4.1 Site Conceptual Model

Figure 4-1 presents a screening level or preliminary ecological site conceptual model (SCM) which details the significant pathways by which site-related contaminants may be transported to other environmental media. The SCM also illustrates the exposure pathways by which ecological receptors may reasonably be exposed to site-related contaminants. Exposure pathways are classified as follows:

- Pathways not complete - Incomplete exposure pathways (i.e., those that are not known to occur) are shown as open boxes and are not evaluated in the SERA.

- Pathways complete but considered insignificant - Exposure pathways considered to be complete but are considered to be insignificant compared to other exposure pathways. These pathways are shown as boxes with vertical hatched lines and are not evaluated in the SERA.

- Pathways complete but risk evaluation impossible - Exposure pathways are complete, but exposure and/or toxicity data are not available to evaluate risks These pathways are shown as boxes with diagonal hatched lines and are not evaluated in the SERA.

- Exposure pathways complete - These exposure pathways are considered to be potentially complete and are evaluated quantitatively in the SERA. These pathways are shown as dark shaded boxes.

The following sections present a more detailed description of sources, transport and migration pathways and exposure pathways for ecological receptors at the RFT Site.

4.1.1 Source Media

As presented in Section 3, contamination exists in several environmental media (surface water, sediment, seep, and soil) at the RFT Site. This contamination originated from a tailings impoundment and other tailings deposits both inside and outside the main impoundment area (Figure 2-1). Currently both the main tailings impoundment and the tailings deposits outside of the impoundment are reported to be covered with a clay soil cover cap (RMC, 2001b). However, recent mapping and sampling data suggest that some of these tailings on and off the impoundment are not uniformly covered. As seen in Table 4-1, soil cover depths for the main impoundment range from 3 inches to 11 feet (RMC, 2001b). Based on arsenic and lead concentrations for the off-impoundment soil samples collected from 0 to 6 inches (Figure 4-2), the observed soil cover is shallow in some areas south of the diversion ditch and absent in other locations. Although these two tailings sources (on and off the impoundment) are separated spatially, the release mechanisms and resulting secondary source medium and exposure media for ecological receptors are generally the same (Figure 4-1).

4.1.2 Migration Pathways (Release Mechanisms)

Contamination in a source medium can migrate and cause contamination in other parts of the environment by pathways that involve either physical transport from one location to another. These transport processes are referred to as release mechanisms. The potential release mechanisms from the source (tailings) to secondary source media and exposure media for ecological receptors are depicted in Figure 4-1. These include historical and current wind erosion, penetration of the soils cap (i.e.: burrowing animals, plant roots), mixing of the cover soils with tailings, infiltration of rainwater and snowmelt, runoff associated with rainwater and snowmelt, and leaching from soils as a result of infiltration of rainwater and snowmelt.

4.1.3 Secondary Source Media

Under dry conditions, particles of either tailings or cover material mixed with tailings can be eroded by wind and transported to adjacent areas resulting in suspended soil/dust/tailings, or contamination of surrounding soil with tailings or a mixture of soil cover and tailings.

The contaminants present in tailings and or soil can be transported by water from surface runoff into surface water bodies (e.g., streams, wetlands and impoundments). This may result in deposition of contaminants absorbed or adsorbed to soil particles as sediments. The dissolved contaminants migrating in runoff water or deposited with sediments may be released to surface waters. Dissolved contaminants in soil may also leach to groundwater, with subsequent transport to surface water as seeps and further possible transfer to surface water or sediments.

Contaminants in surface water, sediment, soil, or seeps can enter the food chain if organisms and plants take up or accumulate contaminants from these media into tissues, which are then consumed by other animals.

4.1.4 Potentially Exposed Receptors and Exposure Pathways

Ecological receptors may be potentially exposed to contaminants in any one of seven exposure media at the RFT Site (Figure 4-1). These exposure media to which ecological receptors may be exposed include suspended soil or dust particles, surface soil/tailings, terrestrial prey items (food chain), sediment, aquatic prey items (food chain), surface water and seeps. The exposure pathways for ecological receptors to contaminants in each of the exposure media are discussed separately in the following subsections.

4.1.4.1 Suspended Soil and Dust

For ecological receptors, exposures to suspended soil and dust can occur via inhalation. Wind erosion of soil can result in the suspension of dust and soil particles into the air which could be inhaled by receptors both on and off the RFT Site. The exposure pathways that are judged to be potentially complete include:

- Inhalation of soil/tailings by birds and mammals
- Inhalation of soil/tailings by amphibians and reptiles

Exposure to suspended soil or dust particles via inhalation is a potentially complete pathway but is generally considered insignificant for wildlife receptors (mammals and birds) in comparison to ingestion exposures. Although airborne soil particulates could be inhaled by wildlife receptor, it is more likely that these respirable particles (>5 um) will be ingested as a result of mucocilliary clearance (Witschi and Last, 1987). These exposures are considered to be quantified through the incidental soil ingestion pathway. For amphibians and reptiles inhalation and ingestion exposures are possible but there is no data available on the toxicity of either inhaled or ingested contaminants to evaluate these pathways.

4.1.4.2 Surface Soil and Tailings

For ecological receptors, exposures to surface soil and tailings can occur via two pathways: direct contact and incidental ingestion. Direct contact with tailings or soil mixed with tailings could occur in areas where the soil cover is thin, where animals burrow through cover soils or where plant roots penetrate the soil cover layer. Terrestrial receptors typically will not intentionally ingest large quantities of soil, however, some incidental ingestion of soil and tailings along with food items does occur (especially in receptors that feed on plants and soil invertebrates). The exposure pathways that are judged to be potentially complete include:

- Direct contact with surface soil/tailings by birds and mammals
- Direct contact with surface soil/tailings by plants and soil invertebrates
- Incidental ingestion of surface soil/tailings by birds and mammals

Dermal exposure to surface soil/tailings is a potentially complete pathway wildlife receptors (mammals and birds) but is generally considered insignificant in comparison to ingestion exposures. For amphibians and reptiles, dermal exposures are possible but there is no data available on the toxicity of dermally applied contaminants to evaluate this pathway. The pathways that are quantitatively evaluated in the SERA are:

- Incidental ingestion of surface soil/tailings by birds and mammals
- Direct contact with surface soil/tailings by plants and soil invertebrates

Analytical data are currently available (see Section 3) for tailings, impoundment cover soils, off-impoundment soils, and background soils for the RFT Site.

4.1.4.3 Terrestrial Food Chain

Contaminants in soils can enter the terrestrial food chain if organisms (i.e.: soil invertebrates, plants and small mammals) take up or accumulate contaminants from soils into tissues, which are then consumed by wildlife receptors. The exposure pathways that are judged to be potentially complete include:

- Ingestion of terrestrial food items by birds and mammals
- Ingestion of terrestrial food items by reptiles

For amphibians and reptiles ingestion exposures to contaminants in the terrestrial food chain are possible but there is no data available on the toxicity of ingested contaminants to evaluate this pathway. The pathways that are quantitatively evaluated in the SERA are:

- Ingestion of terrestrial food items by birds and mammals

Because tissue concentrations are not available for terrestrial food items such as plants, terrestrial or soil invertebrates, or wildlife species, soil concentrations for the RFT Site are used to estimate concentrations in these food items. Use of estimated tissue data rather than measured data is a source of uncertainty in the SERA. This uncertainty is discussed in Section 8 and the lack of terrestrial food chain data is further discussed as a possible data gap in Section 9.

4.1.4.4 Surface Water

Contaminants in surface water may result from the discharge of contaminated groundwater, runoff from the surface soils and tailings, disassociation of contaminants from sediments into surface water and the discharge of contamination from seeps. The exposure pathways that are judged to be potentially complete for contaminants in surface water include:

- Ingestion of surface water by aquatic receptors
- Ingestion of surface water by birds and mammals
- Ingestion of surface water by amphibians and reptiles

- Direct contact with surface water by aquatic receptors
- Direct contact with surface water by birds and mammals
- Direct contact with surface water by amphibians and reptiles

Exposures to contaminants in surface water by ingestion is potentially complete for amphibians, reptiles and aquatic receptors (invertebrates and fish). Data, however, are not available to either estimate toxicity or exposures related to the ingestion pathway for these receptors. Exposures for wildlife receptors (birds and mammals) to contaminants in surface water by dermal contact is potentially complete, but is generally considered insignificant in comparison to ingestion exposures. Exposures to contaminants in surface water by dermal contact is potentially complete for reptiles. Data, however, are not available to either estimate toxicity or exposure for this exposure pathway. The remaining pathways for surface water that are quantitatively evaluated in the SERA are:

- Ingestion of surface water by birds and mammals
- Direct contact with surface water by amphibians
- Direct contact with surface water by aquatic receptors

Analytical data are currently available for surface water for the RFT Site (see Section 3). These data are divided into several surface water exposure locations (units). These include the north and south diversion ditches, the unnamed drainages that flow into the south diversion ditch, ponded water areas, the wetlands area, and Silver Creek.

4.1.4.5 Sediment

Contaminants in sediment may result from the discharge of contaminated groundwater, runoff and erosion from surface soils and tailings, disassociation of contaminants from surface water into sediments and the discharge of contamination from seeps. The exposure pathways that are potentially complete for contaminants in sediment include:

- Incidental ingestion of sediment by aquatic receptors
- Incidental ingestion of sediment by birds and mammals
- Incidental ingestion of sediment by amphibians and reptiles

- Direct contact with sediment by benthic invertebrates
- Direct contact with sediment by birds and mammals
- Direct contact with sediment by amphibians

Exposures to contaminants in sediment by ingestion are potentially complete for amphibians, reptiles and aquatic invertebrates. Data, however, are not available to either estimate toxicity or exposures related to the ingestion pathway for these receptors. Exposures for wildlife receptors (birds and mammals) to contaminants in sediment by dermal contact is potentially complete but is generally considered insignificant in comparison to ingestion exposures. Exposures to contaminants in sediment by dermal contact is potentially complete for reptiles and amphibians. Data, however, are not available to either estimate toxicity or exposure for this exposure pathway. The remaining pathways for surface water that are quantitatively evaluated in the SERA are:

- Incidental ingestion of sediment by birds and mammals
- Direct contact with sediment by benthic invertebrates

Analytical data are currently available for sediment for the RFT Site (see Section 3). These data are divided into several sediment exposure locations that correspond to surface water exposure areas. These include the north and south diversion ditches, the unnamed drainages that flow into the south diversion ditch, ponded water areas, the wetlands area, and Silver Creek..

4.1.4.6 Aquatic Food Chain

Contaminants in surface water and sediment can enter the aquatic food chain if organisms (i.e.: benthic macroinvertebrates, fish, etc.) take up or accumulate contaminants from these media into tissues, which are then consumed by aquatic or wildlife receptors. The exposure pathways that are potentially complete include:

- Ingestion of aquatic food items by birds and mammals
- Ingestion of aquatic food items by aquatic receptors
- Ingestion of aquatic food items by amphibians and reptiles

For amphibians and reptiles ingestion exposures to contaminants in the aquatic food chain are possible but there are no data available on the toxicity of ingested contaminants to evaluate this pathway for these receptors. It is possible to evaluate ingestion exposures for fish to metals in food and sediment. The exposures however are expected to be insignificant compared to direct contact exposures. This exposure pathway will, however, be re-evaluated in the baseline risk assessment as more data becomes available on specific receptors present at the RFT Site. Risks associated with body burdens of contaminants in aquatic organisms (fish) will also be evaluated in the baseline risk assessment if fish tissue residue data becomes available. The pathways that are quantitatively evaluated in the SERA for the aquatic food chain are:

- Ingestion of aquatic food items by birds and mammals

Because tissue concentrations are not available for aquatic food items such as benthic macroivertebrates or fish, sediment concentrations for the RFT Site are used to estimate concentrations in these food items as appropriate. Use of estimated tissue data rather than measured data is a source of uncertainty in the screening assessment; this uncertainty is discussed in Section 8. The lack of aquatic food chain data is further discussed in the data gaps analysis as Section 9.

4.1.4.7 Seeps

To alleviate water pressure at the impoundment, the containment system is constructed to allow water to seep from the impoundment resulting in a seep area located at the toe of the main embankment. Although the flow from the seep is intermittent and low and does not reach Silver Creek via overland flow, it does impact the water chemistry in the wetlands area and it is still a potential exposure location for both aquatic and terrestrial receptors. The exposure pathways to seeps that are potentially complete include:

- Ingestion of seep water by aquatic receptors
- Ingestion of seep water by birds and mammals
- Ingestion of seep water by amphibians and reptiles

- Direct contact with seep water by aquatic receptors
- Direct contact with seep water by birds and mammals
- Direct contact with seep water by amphibians
- Direct contact with seep water by plants

Exposures to contaminants in seep water by ingestion is potentially complete for amphibians, reptiles and aquatic receptors (invertebrates and fish). Data, however, are not available to either estimate toxicity or exposures related to the ingestion pathway for these receptors. Exposures for wildlife receptors (birds and mammals) to contaminants in seep water by dermal contact is potentially complete, but is generally considered insignificant in comparison to ingestion exposures. Exposures to contaminants in seep water by dermal contact is potentially complete for reptiles. Data, however, are not available to either estimate toxicity or exposure for this exposure pathway. The remaining pathways for surface water that are quantitatively evaluated in the SERA are:

- Ingestion of seep water by birds and mammals
- Direct contact with seep water by amphibians
- Direct contact with seep water by aquatic receptors
- Direct contact with seep water by plants

Analytical data from the seep near the main embankment is not currently available. However, it is assumed that seep concentrations are similar to groundwater concentrations measured in wells at the base of the main embankment near the seep.

4.1.5 Changes to Previously Presented Model

The ecological site conceptual model presented as Figure 4-1 is based on site conceptual models presented in the Remedial Investigation SAP (RMC, 2001a - Figures 8a and 8b) with the following additions and changes:

- Separate models were previously presented for on-impoundment and off-impoundment areas. As the exposure pathways and receptors are similar on-impoundment versus off-impoundment these two models were collapsed into one.

- Separate models were previously presented for "upland" versus "wetland" areas. These two areas are still considered in the current model but are not specifically mentioned. It was necessary to elucidate exposure pathways for terrestrial wildlife to both soils in wetland and upland areas as well as surface water and sediments of wetland and stream habitats.

- Potential exposures to receptors to groundwater discharged as seep water and discharged to surface water was added to the ecological site conceptual model.

- The previous models differentiated "potentially significant" pathways from "potential" pathways. The current model identifies both as "potential" pathways. Those "potential" pathways that can be quantified are evaluated in the SERA.

4.2 Selection of Contaminants of Potential Concern

Contaminants of Potential Concern (COPCs) are contaminants which exist in the environment at concentrations that might be of potential concern to ecological receptors, and which are derived, at least in part, from site-related sources. Exposure pathways and media of concern for ecological receptors are identified and presented in the SCM (Figure 4-1). These exposure pathways and media of concern provide the assumptions for evaluating the appropriate media and receptors in the SERA. The purpose of the COPC selection procedure is to eliminate contaminants that are clearly not of potential ecological concern, and to carry forward those contaminants that might be of concern. The principal steps in eliminating or retaining a contaminant as an ecological COPC are described in Section 4.2.1 and are depicted in Figure 4-3. The results of the screening process are described in Section 4.2.2.

4.2.1 Screening Steps

4.2.1.1 Eliminate Contaminants Never Detected

In accord with USEPA (1989), a contaminant is a candidate for elimination from the quantitative risk assessment if it is detected infrequently or if there is no reason to believe that the contaminant may be present (i.e., when a contaminant is not site-related). Using this logic, a contaminant never detected in a media is eliminated from the quantitative risk assessment.

For contaminants that have never been detected, it is important to evaluate the adequacy of the detection limits for the available data. If the maximum detection limit for a contaminant is above available toxicity benchmarks, it should be evaluated qualitatively and identified as a source of uncertainty. It is assumed that these contaminants would only have a negligible effect on risk levels and would not likely result in a significant underestimate of risk.

4.2.1.2 Retain Contaminants Detected that are Bioaccumulative

Contaminants considered to be bioaccumulative are retained as COPCs if they are detected regardless of frequency of detection. Bioaccumulative contaminants of concern (BCCs) are defined as part of the Great Lakes Water Quality Guidance (GLWQG) wildlife Tier I criteria. There are 22 listed BCCs, of which one contaminant --mercury-- is detected at the RFT Site. Therefore, mercury is retained as a COPC. There are no other detected contaminants that are defined as bioaccumulative.

4.2.1.3 Eliminate Contaminants Detected Infrequently

In accord with USEPA (1989), a contaminant is a candidate for elimination from the quantitative risk assessment if it is detected infrequently. If a contaminant is detected infrequently (detection frequency is less than five percent), the contaminant is considered to be of little concern, but is evaluated qualitatively and identified as a source of uncertainty.

4.2.1.4 Eliminate Contaminants that are Considered to be Physiological Electrolytes

Several of the analytes measured in environmental media are considered to be essential physiological electrolytes for birds, mammals, plants and/or soil invertebrates. These analytes are eliminated as COPCs and include calcium, iron, magnesium, potassium, and sodium. Physiological electrolytes are not carried forward in the SERA.

4.2.1.5 Eliminate Contaminants Detected at Concentrations less than Background

This step involves comparing site contaminant concentrations to reference or background concentrations. Background for the purposes of the SERA are upgradient (upstream) concentrations of metals; those concentrations that do not represent contamination from the site. It is important to note that these samples are representative of anthropogenic, non-site related levels, they do not represent "pristine" (not influenced by human activity) environmental levels. In instances where the number of samples (N) is less than five, the reference data set is considered to be too small and a reference comparison is not made.

For the RFT Site, soil background samples were collected from eleven areas surrounding the site identified as not affected by wind-blown tailings. However, most (9 of 11) samples were only analyzed for arsenic and lead, and only two samples were analyzed for all RCRA metals. In addition, although sampling locations were selected from areas thought not to be affected by tailings, sampling location BG11 was later found to have been inadvertently placed near tailings. Because of the limited number of samples, limited number of analytes and the uncertainty in the representativeness of the data as "background", the background comparison screening step is not included as part of the COPC screening process for the SERA.

4.2.1.6 Eliminate Contaminants with Maximum Concentrations less than an Established Level of Concern

This step involves comparing the maximum detected contaminant concentration in an exposure medium to an appropriate ecologically-based screening level. If the maximum detected value is less than the screening level, the contaminant does not pose a potential risk and is eliminated as a COPC. If no ecologically-based screening level is available, the constituent is retained as a COPC. Separate screening processes are completed for aquatic and terrestrial receptors, resulting in two separate lists of COPCs.

COPC Selection Process for Aquatic Receptors. Surface water screening benchmarks for aquatic receptors are based on chronic ambient water quality criteria (AWQC) for both dissolved and total recoverable metals. AWQC values are derived from data for a wide range of aquatic species, and are intended to protect at least 95% of aquatic receptor (benthic invertebrate, plant, and fish) species from unacceptable adverse effects. Sediment screening benchmarks for benthic invertebrates are identified from Ingersoll et al. (1996) and Long and Morgan (1991). Screening benchmarks for surface water and sediment are listed in Table 4-2.

COPC Selection Process for Terrestrial Wildlife. Terrestrial wildlife screening benchmarks were identified from Sample et al. (1996), Pedigo et al. (1988), and Skorupa (1998). These benchmarks represent contaminant concentrations in drinking water and diet that are not expected to be associated with adverse effects to wildlife species. The screening benchmarks derived by Sample et al. (1996) are presented for 20 wildlife species. The lowest benchmark concentrations were selected for use in the screening process. Drinking water benchmarks were used to screen surface water data, while the dietary benchmarks were used to screen sediment and soil data. The use of the dietary benchmarks for sediment and soil screening is conservative, as the rate of incidental ingestion by wildlife is expected to be much lower than that for the diet. These screening benchmarks are summarized in Table 4-3.

4.2.2 Application of COPC Selection Methodology

4.2.2.1 Surface Water

The available surface water data are discussed in Section 3. The surface water data set includes samples from the south diversion ditch, the unnamed drainages that flow into the south diversion ditch, ponded water areas, and Silver Creek (Figure 2-1). Tables 4-4 and 4-5 summarize the COPC selection for surface water at the RFT Site for aquatic and terrestrial receptors, respectively. As seen, the left side of each table lists for each of the analytes: the number of detections, the number of samples, the detection frequency, and the mean and maximum concentrations for non-detects and detects.

COPCs for Aquatic Receptors. The results of the surface water COPC selection process for aquatic receptors are summarized in Table 4-4 for dissolved and total recoverable metals. Seventeen contaminants are identified as COPCs in surface water for aquatic receptors including aluminum, antimony, arsenic, barium, beryllium, boron, cadmium, chromium, cobalt, copper, cyanide, lead, manganese, mercury, selenium, silver and zinc. Potential risks for aquatic receptors associated with these COPCs are evaluated further in the risk characterization sections of this SERA.

COPCs for Terrestrial Receptors. Table 4-5 provides the results of the surface water COPC selection process for terrestrial receptors. Six contaminants are identified as COPCs in surface water for terrestrial wildlife receptors: arsenic, lead, mercury, selenium, silver and zinc. Potential risks for this COPC are evaluated further in the risk characterization sections of this SERA.

4.2.2.2 Sediment

The available sediment data are discussed in Section 3. The sediment data set includes samples from the south diversion ditch, the wetland area, and Silver Creek (Figure 2-1). Tables 4-6 and 4-7 summarize the COPC selection for sediments at the RFT Site for aquatic and terrestrial receptors, respectively. As seen, the left side of each table lists for each of the analytes: the number of detections, the number of samples, the detection frequency, and the mean and maximum concentrations for non-detects and detects.

COPCs for Benthic Invertebrates. The results of the sediment COPC selection process for benthic invertebrates are summarized in Table 4-6. Eighteen contaminants are identified as COPCs in sediment for aquatic receptors, including aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, lead, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc. Potential risks for these COPCs are evaluated further in the risk characterization sections of this SERA.

COPCs for Terrestrial Receptors. Table 4-7 provides the results of the sediment COPC screen for terrestrial receptors. Seventeen contaminants are identified as COPCs in sediment for terrestrial wildlife receptors, including aluminum, antimony, arsenic, barium, cadmium, chromium, cobalt, copper, lead, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc. Potential risks for these COPCs are evaluated further in the risk characterization sections of this SERA.

4.2.2.3 Soils and Tailings

The available data sets for tailings and soils are discussed in Section 3. Site tailings, cover soils (both on and off the impoundment), and the background soils were combined into one data set for the purposes of the COPC screen. Table 4-8 summarizes the COPC selection for soils and tailings at the RFT Site for terrestrial receptors. As seen, the left side of the table lists for each of the analytes: the number of detections, the number of samples, the detection frequency, and the mean and maximum concentrations for non-detects and detects.

COPCs for Terrestrial Receptors. Table 4-8 provides the results of the soils and tailings COPC screen for terrestrial receptors. Twelve contaminants are identified as COPCs in soils and tailings for terrestrial wildlife receptors, including aluminum, antimony, arsenic, barium, cadmium, chromium, copper, lead, mercury, selenium, silver, and zinc. Potential risks for these COPCs are evaluated further in the risk characterization sections of this SERA.

4.2.3 Summary

The exposure pathways selected for quantitative evaluation in the SERA including the following:

Aquatic Receptors

- Direct contact with surface water and seep water for fish and benthic invertebrates
- Direct contact with sediments by benthic invertebrates

Amphibians

- Direct contact with surface water and seep water

Birds & Mammals

- Ingestion of surface water and seep water
- Ingestion of terrestrial and aquatic food items
- Incidental ingestion of sediment and soil and/or tailings

Terrestrial Plants & Soil Fauna

- Direct contact with soil and/or tailings
- Direct contact with seep water

The COPCs selected for each of these exposure pathways and media of concern based on the SCM (Figure 4-1) are summarized in the following text table:

Summary of COPCs Selected for Evaluation in the SERA					
Analyte	Surface Water		Sediment		Soil & Tailings
	Aquatic Receptors	Terrestrial Receptors	Aquatic Receptors	Terrestrial Receptors	Terrestrial Receptors
Aluminum	X		X	X	X
Antimony	X		X	X	X
Arsenic	X	X	X	X	X
Barium	X		X	X	X
Beryllium	X		X		
Boron	X				
Cadmium	X		X	X	X
Chromium	X		X	X	X
Cobalt	X		X	X	
Copper	X		X	X	X
Cyanide	X				
Lead	X	X	X	X	X
Manganese	X		X	X	
Mercury	X	X	X	X	X
Nickel			X	X	
Selenium	X	X	X	X	X
Silver	X	X	X	X	X
Thallium			X	X	
Vanadium			X	X	
Zinc	X	X	X	X	X
Total COPCs	*17*	*6*	*18*	*17*	*12*

4.3 Identification of Assessment and Measurement Endpoints

4.3.1 Identified Goals for the Screening Ecological Risk Assessment

The overall management goal for ecological health at the RFT Site is stated as the following:

> *Ensure adequate protection of ecological systems within the impacted areas of the Richardson Flat Tailings Site by protecting them from the deleterious effects of acute and chronic exposures to site-related contaminants of concern.*

In order to provide specificity regarding this general goal and identify specific measurable ecological values to be protected, the following list of sub-goals was derived:

- Ensure adequate protection of terrestrial soil fauna and plant communities, including native plant communities, by protecting them from the deleterious effects of acute and chronic exposures to site-related contaminants of concern.

- Ensure adequate protection of aquatic and amphibian life in Silver Creek, the site diversion ditches and wetlands areas from the deleterious effects of acute and chronic exposures to site-related contaminants of concern.

- Ensure adequate protection of terrestrial mammal and bird populations by protecting them from the deleterious effects of acute and chronic exposures to site-related contaminants of concern.

- Ensure adequate protection of threatened and endangered species (including candidate species) and species of special concern and their habitat by protecting them from the deleterious effects of acute and chronic exposures to site-related contaminants of concern.

(Note: "Adequate" protection is generally defined as protective of growth, reproduction, and survival of local populations.)

4.3.2 Identification of Assessment and Measurement Endpoints

Assessment endpoints are explicit statements of the characteristics of the ecological system that are to be protected. Assessment endpoints are either measured directly or are evaluated through indirect measures. Measurement endpoints represent quantifiable ecological characteristics that can be measured, interpreted, and related to the valued ecological components chosen as the assessment endpoints (USEPA, 1992; 1997).

The following assessment and measurement endpoints are used to interpret potential ecological risks for the RFT Site for the SERA. In some cases, more than one measurement endpoint is identified for a particular assessment endpoint. These instances permit a weight-of-evidence approach to be used in risk characterization. In other cases, a measurement endpoint may be relevant to more than one assessment endpoint.

Assessment Endpoint	Measurement Endpoint
Protection of terrestrial plants and soil fauna from adverse effects related to exposure to COPCs in surface soil.	Comparison of COPC concentrations in soil to terrestrial toxicity benchmarks.
Protection of benthic invertebrates, fish and amphibians from adverse effects related to exposure to COPCs in surface water and sediment.	Comparison of sampling location-specific COPC concentrations in surface water and sediment to aquatic toxicity benchmarks.
Protection of terrestrial wildlife from adverse effects to growth, reproduction or survival related to exposure to COPCs in surface water, sediment, soil, and food items.	Comparison of the predicted average daily doses of COPCs from surface water, sediment, and food to toxicity reference values.

5.0 SCREENING LEVEL EXPOSURE ASSESSMENT

5.1 Aquatic Receptors

5.1.1 Surface Water

Aquatic receptors (benthic invertebrates, plants, fish and amphibians) are potentially exposed to COPCs in surface water via direct contact. The exposure point concentration (EPC) for aquatic receptors to COPCs in surface water is either the 95th upper confidence limit (95UCL) of the mean or the maximum concentration, whichever is lower. For some locations, limited samples are available; at these locations the EPC is usually equal to the maximum measured concentration. COPCs that are non-detects (U qualified; below the detection limit) are evaluated at one-half the reported detection limit in the calculation of the EPC. For the purposes of the SERA, direct contact exposures with surface water are evaluated on a sampling location-specific basis. The location specific EPCs for each COPC by sampling location are listed in Table 5-1. These EPCs are compared to toxicity benchmarks identified in Section 6.1.1 for benthic invertebrates and fish and Section 6.2 for amphibians to identify potential risks for each, respectively in Section 7.1.1 and 7.2.1.

5.1.2 Sediment

Benthic invertebrates are potentially exposed to COPCs in sediment via direct contact. The EPC for benthic invertebrates to COPCs in sediments is either the 95th upper confidence limit (95UCL) of the mean or the maximum concentration, whichever is lower. For some locations, only one or a limited number of samples are available; therefore the EPC is usually equal to the maximum measured concentration. COPCs that are non-detects (U qualified; below the detection limit) are evaluated at one-half the reported detection limit in the calculation of the EPC. For the purposes of the SERA, direct contact exposures with sediment are evaluated on a sampling location-specific basis. The location specific EPCs for sediment for each COPC by sampling location are listed in Table 5-2. These EPCs are compared to toxicity benchmarks identified in Section 6.1.2 to identify potential risks for aquatic receptors in Section 7.1.1.2.

5.1.3 Seeps

Benthic invertebrates and amphibians are potentially exposed to COPCs in seep water via direct contact. The EPC for benthic invertebrates and amphibians to COPCs in seep water is either the 95th upper confidence limit (95UCL) of the mean or the maximum concentration, whichever is lower. COPCs that are non-detects (U qualified; below the detection limit) are evaluated at one-half the reported detection limit in the calculation of the EPC. For the purposes of the SERA, direct contact exposures with seep water are evaluated for each monitoring well (groundwater data used to estimate seep concentrations). The EPCs for each COPC by monitoring well are listed in Table 5-3. These EPCs are compared to toxicity benchmarks identified in Section 6.1.1 for benthic invertebrates and fish and Section 6.2 for amphibians to identify potential risks for each, respectively in Section 7.1.3 and 7.2.2 .

5.2 Terrestrial Plants and Soil Fauna

5.2.1 Soils

Terrestrial plants and soil invertebrates are potentially exposed to COPCs in soils via direct contact. Exposures for these receptors are evaluated on a sampling location-specific basis. The EPC for plants and soil invertebrates is equal to the average concentration across all depths at each sampling location for each COPC. The EPCs are listed for each soil sampling location in Appendix F. The EPC for each COPC for each sampling location is compared to toxicity benchmarks for terrestrial plants and soil invertebrates presented in Sections 6.3 and 6.4, respectively, to identify potential risks for these receptors from direct contact with COPCs in soil in Sections 7.3 and 7.4.

5.2.2 Seeps

Terrestrial plants are potentially exposed to COPCs in seeps via direct contact. Exposures are evaluated for each monitoring well used to estimate seep water concentrations. The EPC for each COPC in seep water (groundwater) is equal to the 95th upper confidence limit (95UCL) of the mean or the maximum concentration, whichever is lower. COPCs that are non-detects (U qualified; below the detection limit) are evaluated at one-half the reported detection limit in the calculation of the EPC. The EPCs are listed for each groundwater well in Table 5-3. The EPC for each COPC for each sampling location is compared to aqueous toxicity benchmarks for terrestrial plants in Sections 6.3.2 to identify potential risks for plants exposed to COPCs in seep water in 7.3.2.

5.3 Wildlife

Wildlife species may be exposed to COPCs by ingestion of surface water, seep water, sediments,, soils and food items that have taken up contaminants into their tissues. Exposures for wildlife receptors to each environmental medium of concern are assessed for five exposure areas at the RFT Site (Figure 2-1) including:

- Upstream Silver Creek,
- Downstream Silver Creek,
- The south diversion ditch,
- Ponded water areas on the impoundment, and
- Unnamed drainages which flow into the south diversion ditch.

The following subsections describe how wildlife species are selected for evaluation and how COPC exposure doses are estimated for wildlife for each exposure medium for each exposure area.

5.3.1 Identification of Representative Wildlife Species

It is not feasible to evaluate exposures and risks for each avian and mammalian species potentially present within the study area. For this reason, specific wildlife species are identified as representative wildlife species for the purpose of estimating quantitative exposures (doses) in the SERA. The representative species are wildlife species that are potentially present within the Site area and are representative of other species with similar dietary preferences and feeding guilds. Selection criteria for representative wildlife species includes trophic level, feeding habits, and the availability of life history information. Representative wildlife receptors selected for the RFT Site are summarized in the following text table.

Summary of Representative Wildlife Receptors		
Type	**Species**	**Represents**
Small Mammalian Omnivores	Deer Mouse (*Peromyscus maniculatus*)	Small mammalian terrestrial omnivore receptors ingesting terrestrial food items (vegetation & terrestrial invertebrates), soil, and surface water.
Small Mammalian Insectivores	Masked Shrew (*Sorex cinereus*)	Small mammalian terrestrial insectivore receptors ingesting terrestrial food items (soil invertebrates), soil, and surface water.
Mammalian Carnivores	Red Fox (*Vulpes vulpes*)	Mammalian carnivore receptors ingesting terrestrial food items (small mammals), soil, and surface water.
Mammalian Piscivores	Mink (*Mustela vison*)	Mammalian piscivore receptors ingesting aquatic food items (fish), sediment, and surface water.
Small Avian Insectivores	Mallard Duck (*Anas platyrhynchos*)	Avian insectivore receptors ingesting aquatic food items (benthic invertebrates), sediment, and surface water.
Small Avian Herbivores	Greater-Sage Grouse (*Centrocercus urophasianus*)	Small avian terrestrial herbivore receptors ingesting terrestrial food items (vegetation), soil, and surface water.
Small Avian Omnivores	American Robin (*Turdus migratorius*)	Avian omnivore receptors ingesting terrestrial food items (vegetation & soil invertebrates), soil, and surface water.
Avian Carnivores	American Kestrel (*Falco sparverius*)	Avian carnivore receptors ingesting terrestrial food items (small mammals), soil, and surface water.
Avian Piscivores	Belted Kingfisher (*Ceryle alcyon*)	Avian piscivore receptors ingesting aquatic food items (fish), sediment, and surface water.

Some species-specific factors are needed to estimate doses of COPCs including body weight, ingestion rates, and dietary composition. These wildlife exposure factors are derived largely from the Wildlife Exposure Factors Handbook (USEPA, 1993a and b). The exposure factors including derivation and sources are provided as Appendix B. A summary of the exposure factors selected for the selected wildlife receptors is provided in Table 5-4.

5.3.2 Estimation of Doses Associated with Ingestion of Surface Water or Seep Water

Exposures to COPCs in surface water are quantified based on the following equation:

$$Dose_{sw} = \frac{IR_{sw} x C_{sw}}{BW} xAUF$$

where:

IR_{sw}	=	Ingestion rate of surface water or seep water for the receptor of interest (L/day);
C_{sw}	=	Concentration of COPC in sediment (mg/L);
AUF	=	Area Use Factor; and
BW	=	Body weight of the receptor of interest (kg wet weight).

C_{sw} is equal to the EPC of each COPC for surface water within each exposure area. The EPC is equal to either the 95th upper confidence limit (95UCL) of the mean or the maximum concentration, whichever is lower. COPCs that are non-detects (U qualified; below the detection limit) are evaluated at one-half the reported detection limit in the calculation of the EPC. The surface water EPC concentrations for each COPC by exposure area are listed in Table 5-5. These EPC concentrations are compared to toxicity reference values (TRVs) calculated for wildlife in Section 6.5 to estimate risks for wildlife for ingestion of COPCs in surface water in Section 7.5.1 and seep water in Section 7.5.3. The AUF for each wildlife species is conservatively assumed to be 100%.

5.3.3 Estimation of Doses Associated with Ingestion of Sediments

Exposures to COPCs in sediment are quantified based on the following equation:

$$Dose_{sed} = \frac{IR_{sed} x C_{sed}}{BW} xAUF$$

where:

IR_{sed}	=	Ingestion rate of sediment for the receptor of interest (kg dry weight/day);
C_{sed}	=	Concentration of COPC in sediment (mg/kg dry weight);
AUF	=	Area Use Factor; and
BW	=	Body weight of the receptor of interest (kg wet weight).

C_{sed} is equal to the EPC for each COPC for sediment within each exposure area. The EPC is equal to either the 95th upper confidence limit (95UCL) of the mean or the maximum concentration, whichever is lower. COPCs that are non-detects (U qualified; below the detection limit) are evaluated at one-half the reported detection limit in the calculation of the EPC. The sediment EPC concentrations for each COPC by exposure area are listed in Table 5-6. These EPC concentrations are compared to toxicity reference values (TRVs) calculated for wildlife in Section 6.5 to estimate risks for wildlife for ingestion of COPCs in sediment in Section 7.5.2. The AUF for each wildlife species is conservatively assumed to be 100%.

5.3.4 Estimation of Doses Associated with Ingestion of Soils/Tailings

Exposures to COPCs in soil/tailings are quantified based on the following equation:

$$Dose_{sed} = \frac{IR_{sed} x C_{sed}}{BW.} xAUF$$

where:

IR_{soil}	=	Ingestion rate of soil for the receptor of interest (kg dry weight/day);
C_{soil}	=	Concentration of COPC in soil (mg/kg dry weight);
AUF	=	Area Use Factor; and
BW	=	Body weight of the receptor of interest (kg wet weight).

C_{soil} is equal to the EPC of each COPC for soil/tailings at each exposure area. The AUF for each wildlife species is conservatively assumed to be 100%. The estimated doses for exposure to COPCs in soil/tailings are calculated for each representative wildlife species and presented in Section 6. The estimated doses are compared to dietary ingestion TRVs in Section 6.2 to characterize risks.

C_{sed} is equal to the EPC for each COPC for soil within each exposure area. The EPC is equal to either the 95th upper confidence limit (95UCL) of the mean or the maximum concentration, whichever is lower. COPCs that are non-detects (U qualified; below the detection limit) are evaluated at one-half the reported detection limit in the calculation of the EPC. The soil EPC concentrations for wildlife for each COPC by exposure area are listed in Table 5-7. These EPC concentrations are compared to toxicity reference values (TRVs) calculated for wildlife in Section 6.5 to estimate risks for wildlife for ingestion of COPCs in soil in Section 7.5.4. The AUF for each wildlife species is conservatively assumed to be 100%.

5.3.5 Estimation of Doses Associated with Ingestion of Food Items

Dietary exposures are possible for terrestrial wildlife by ingestion of terrestrial food chain items (soil invertebrates, plants, birds and mammals) and/or ingestion of aquatic food chain items (plants, benthic invertebrates, and fish). For the SERA, five food types are included in the wildlife exposure model including aquatic invertebrates, fish, terrestrial vegetation, soil invertebrates and small mammals.

The dietary intake of a COPC for each representative species is estimated by the following equation:

$$Dose_{diet} = \frac{IR_{food} \times \sum(C_{food\,i} \times df_i)}{BW}$$

IR_{food}	=	Ingestion rate of food for the receptor of interest (kg dry weight/day);

C_{foodi} = Concentration of COPC in food type "i" (aquatic invertebrate, fish, plant or soil invertebrate) (mg/kg wet weight);
df_i = Dietary fraction (proportion in the diet) of food type "i" (unitless) for the receptor of interest;
BW = Body weight for the receptor of interest (kilograms).

For the SERA, measured biological tissue data is not available; therefore, the calculation of dietary exposure concentrations and doses for wildlife receptors is based on estimated tissue concentrations using bioaccumulation factors (BAFs) for each COPC for each media of concern. C_{food} is equal to the estimated concentration of each COPC in biota within each exposure area. The estimated concentrations of COPCs in food items are based on the EPC concentrations in the respective environmental media (surface water, sediment or soil). The EPC concentrations in food items are listed in Table 5-8. The following subsections describe how concentrations of COPCs in food items are estimated and doses for wildlife calculated for each food item.

5.3.5.1 Benthic Invertebrates and Fish

In order to evaluate food chain exposures for terrestrial wildlife consuming aquatic receptors (benthic invertebrates and fish) at the RFT Site it is necessary to estimate tissue concentrations. Metal tissue concentrations in benthic invertebrates are estimated using equations that estimate the bioaccumulation of inorganic elements into freshwater invertebrate tissues from sediment. These biota-sediment accumulation factors (BSAFs) focus primarily on invertebrates with terrestrial adult stages (i.e.: mayflies) or are prey items for fish (i.e.: amphipods, tubificid worms) and are intended for use in screening level ecological risk assessments to determine the need for further evaluation (BJC, 1998). Based on the model recommendations, the 90th percentile BSAF based on both depurated and non-depurated organisms is used to derive benthic tissue concentrations from sediment.

Parameter	90th Percentile BSAF
Arsenic	0.69
Cadmium	41.55
Chromium	0.468
Cobalt	5.25
Copper	23.87
Lead	0.607
Mercury	2.868
Nickel	2.32
Zinc	7.527

[conc in benthic dw] = BSAF * [conc in sediment dw]

Estimated tissue concentrations in benthic invertebrates, based on sediment EPC concentrations, are calculated for each exposure area in Appendix C. A summary of these concentrations are provided in Table 5-8. These concentrations are used to estimate doses for wildlife consuming benthic invertebrates.

The doses are provided in Appendix C. The doses are compared to TRVs from Section 6.5 to characterize risks for wildlife receptors from the ingestion of benthic invertebrates in Section 7.5.5.1.

Metal tissue concentrations in fish tissue are assumed, conservatively, to be equal to sediment concentrations. This is assumed to represent both uptake from surface water and sediments. The actual extent of bioaccumulation of metals from surface water and sediments into fish tissue is dependant on multiple site-specific factors that are difficult to model.

Estimated tissue concentrations in fish, based on sediment EPC concentrations are calculated for each exposure area in Appendix C. A summary of these concentrations are provided as Table 5-8. These concentrations are used to estimate doses for wildlife consuming fish. The doses are provided in Appendix C. The doses are compared to TRVs from Section 6.5 to characterize risks for wildlife consuming fish in Section 7.5.5.2.

5.3.5.2 Terrestrial Plants

In order to evaluate food chain exposures for wildlife consuming terrestrial plants, plant tissue concentrations are estimated for each exposure area using equations that estimate the bioaccumulation of inorganic elements into terrestrial plant tissues based on soil concentrations. Bechtel Jacobs Company (BJC) (1998) reviewed available literature for collocated soil and plant data to derive empirical models for the uptake of metals from soil to plants. BJC (1998) concluded that for ecological risk assessments, a single-variable regression model better estimates plant tissue concentrations from soil concentrations than use of a single uptake factor. For several inorganic elements (such as cadmium, mercury, selenium, and zinc), a multiple regression model that includes pH is preferred. Unfortunately, data regarding soil pH is not available at the RFT Site, therefore all plant tissue estimates are calculated using the single-variable regression model.

Parameter	B_0	B_1	R^2
Arsenic	-1.992	0.564	0.145
Cadmium	-0.476	0.546	0.447
Copper	0.669	0.394	0.314
Mercury	-0.996	0.544	0.598
Lead	-1.328	0.561	0.243
Selenium	-0.678	1.104	0.633
Zinc	1.575	0.555	0.402

$\ln(\text{plant}) = B_0 + B_1 * \ln(\text{soil})$

where all concentrations are expressed as mg/kg dw

Estimated tissue concentrations of COPCs in plants based on soil EPC concentrations are calculated in Appendix C. A summary of these concentrations are provided in Table 5-8. These concentrations are used to estimate doses for wildlife consuming plants. The doses are provided in Appendix C. These doses are compared to TRVs from Section 6.5 to characterize risks for wildlife receptors from the ingestion of plants in Section 7.5.5.3.

5.3.5.3 Terrestrial Invertebrates (Earthworms)

In order to evaluate food chain exposures from soil invertebrates, earthworm tissue concentrations are estimated for each exposure area using bioaccumulation models derived by Sample et al. (1998a). Sample et al. (1998a) developed a database of soil and earthworm tissue concentrations for several inorganic and organic chemicals based on 32 studies from 11 countries and 5 states. For almost all inorganic elements, a single-variable regression model provides the best estimates of earthworm tissue concentrations. For cadmium and lead, a multiple regression model including soil calcium improved the model fit. Measured data regarding soil calcium, however is not available for most soil samples collected at the RFT Site, therefore all earthworm tissue estimates are calculated using the single-variable regression model. No model is identified to accurately predict chromium or nickel concentrations in earthworm tissue.

Parameter	B_0	B_1	R^2
Arsenic	-1.421	0.706	0.26
Cadmium	2.114	0.795	0.67
Copper	1.675	0.264	0.18
Mercury [a]	0.0781	0.3369	0.51
Lead	-0.218	0.807	0.8
Selenium [b]	-0.075	0.733	0.43
Zinc	4.449	0.328	0.45

$\ln(\text{earthworm}) = B_0 + B_1 * \ln(\text{soil})$
where all concentrations are expressed as mg/kg dw
[a] Based on model data only, validation data excluded
[b] Based on data set with outlier excluded

Tissues concentrations of COPCs in earthworms are estimated for each exposure area based on the EPC values for soil. The calculations are provided as Appendix C and the results are summarized in Table 5-8. These concentrations are used to calculate doses for wildlife species consuming soil invertebrates for each exposure area. These calculations are provided in Appendix C. The doses are compared to TRVs calculated in Section 6.5 to estimate risks for wildlife consuming soil invertebrates in Section 7.5.5.4.

5.3.5.4 Small Mammals

In order to evaluate food chain exposures for wildlife species consuming small mammals, tissue concentrations are estimated for each exposure area using bioaccumulation models derived by Sample et al. (1998b). Sample et al. (1998b) developed a database of soil and small tissue concentrations for 14 inorganic and 2 organic chemicals based on 20 different studies. Small mammal species are divided into 3 trophic feeding groups based on diet – herbivore, insectivore, and omnivore. If sufficient data were available for each trophic group (N>4), trophic-group-specific regression models were developed based on whole body tissue concentrations. If there was insufficient data or if trophic-group-specific models were not reliable, general regression models, which included all trophic group data were developed. For most inorganic elements, a single-variable regression model was used to estimate small mammal tissue concentrations. For barium and mercury in all trophic groups and for chromium and copper in herbivores, the estimated tissue concentration was based on the median uptake factor.

Parameter	Trophic Group	Equation used for Estimation	B_0	B_1	Median Uptake Factor	R^2
Arsenic	Insectivore	General	-4.8471	0.8188	--	0.52
	Herbivore	Trophic-group regression	-5.6531	1.1382	--	0.72
	Omnivore	Trophic-group regression	-4.5796	0.7354	--	0.41
Barium	All	Median general UF	–	–	0.0168	--
Cadmium	Insectivore	Trophic-group regression	0.815	0.9638	--	0.53
	Herbivore	Trophic-group regression	-1.2571	0.4723	--	0.64
	Omnivore	Trophic-group regression	-1.5383	0.566	--	0.63
Chromium	Insectivore & Omnivore	General	-1.4599	0.7338	--	0.42
	Herbivore	Median trophic group UF	–	–	0.0774	--
Copper	Insectivore	Trophic-group regression	2.1042	0.1783	--	83
	Herbivore	Median trophic group UF	–	–	0.0525	--
	Omnivore	Trophic-group regression	1.4592	0.2681	--	0.48
Mercury	All	Median general UF	–	–	0.0543	--
Lead	Insectivore	Trophic-group regression	0.4819	0.4869	--	0.53
	Herbivore	Trophic-group regression	-0.6114	0.5181	--	0.68
	Omnivore	General	0.0761	0.4422	--	0.37
Selenium	All	General	-0.4158	0.3764	--	0.31
Zinc	All	General	4.4713	0.0738	--	0.13

$\ln(\text{small mammal}) = B_0 + B_1 * \ln(\text{soil})$
small mammal = median uptake factor * soil
where all concentrations are expressed as mg/kg dw

Tissue concentrations of each COPC for each exposure area are estimated based on the soil EPC values. The calculations are provided in Appendix C. A summary of the concentrations by exposure area are listed in Table 5-8. These concentrations are used to estimate doses for wildlife consuming small mammals. The calculations are provided in Appendix C. The doses are compared to TRV values calculated in Section 6.5 to estimate risks for wildlife consuming small mammals in Section 7.3.5.5.

6.0 SCREENING LEVEL EFFECTS ASSESSMENT

Potential risks for ecological receptors are estimated in the SERA based on the Hazard Quotient (HQ) approach. The exposure concentrations (or doses) identified in Section 5 are compared to respective toxicity screening benchmarks to calculate an HQ value. If the HQ is less than or equal to one, then no potential for adverse effects is expected. If the HQ exceeds one, adverse effects are possible. This section identifies the toxicity screening benchmarks for each receptor for each exposure medium.

6.1 Toxicity Benchmarks for Aquatic Receptors

6.1.1 Screening Benchmarks for Surface Water and Seeps

The USEPA has derived acute 24-hour and chronic 4-day Ambient Water Quality Criteria (AWQC) values for a number of metals in surface water, including each of the metals of potential ecological concern at the RFT Site (USEPA, 1985b-e; USEPA, 1987; USEPA, 1996; USEPA, 2001b). These AWQC values are based on thorough review of available toxicological information and toxicity testing on the effects of the metal on aquatic receptors (including benthic invertebrates, fish, and aquatic plants), and each criterion is intended to protect 95% of the aquatic genera for which toxicity data are available (USEPA, 1985a).

An important characteristic of AWQC values is that many (but not all) depend on the properties of the test water, especially hardness. Thus, the AWQC for many metals are not fixed values, but increase as hardness increases. The generic form of the equation used to calculate the AWQC (expressed in units of ug/L) at a given hardness H (expressed in units of mg/L) is as follows:

$$AWQC_{total} = \exp[a \times \ln(H) + b]$$

The parameters a and b are empirically-derived coefficients of the best fit straight line through the data in log space. That is:

$$\ln(AWQC_{total}) = a \times \ln(H) + b$$

In cases where the value of AWQC does not depend on hardness (e.g., arsenic), the value of 'a' is zero and the equation reduces to:

$$AWQC_{total} = \exp(b) = \text{Constant}$$

Originally, all AWQC are expressed in terms of total recoverable metal, and are used by comparison to the total recoverable metal concentrations measured in surface water at the site. Subsequently, the EPA concluded that dissolved metals (rather than total metals) are a better indicator of potential risks due to direct contact (e.g., gill respiration in fish) as this concentration represents the amount of the constituent that is biologically available (USEPA, 1995). As a result, the EPA has identified a method for adjusting the AWQC based on total metals which is suitable for use in evaluating risks from dissolved metals

(USEPA 1995). The general form of the equation used to adjust the criterion from total to dissolved is as follows:

$$AWQC_{dissolved} = AWQC_{total} \times \text{Conversion Factor}$$

$$\text{Conversion Factor} = m - n \times \ln(H)$$

The parameters m and n are empirically-derived coefficients of the equation relating total and dissolved concentrations of the metal in laboratory water.

In some cases the conversion factor does not depend on hardness (e.g., arsenic, copper, zinc), so the value of 'n' is zero and the equation reduces to:

$$\text{Conversion Factor} = m$$

However, evaluation of risks to receptors based only on dissolved metal levels could tend to underestimate the total risk across all exposure pathways, including direct contact with solids (either as sediment or suspended in the river) as well as ingestion of contaminated foods and sediments. Even though total recoverable metal levels in surface water may not correlate well with risks from direct contact exposure, use of this more conservative concentration value can help compensate for the omission of risks from other exposure pathways.

Table 6-1 summarizes the parameters (a, b, m, n) needed to calculate the acute and chronic default AWQC for total and dissolved metals of potential concern at the RFT Site and presents AWQC values for each metal at a hardness of 100 mg/L. Also presented are the specified hardness limits for derivation of the AWQC, if the measured station hardness is outside of the specified hardness limits, the applicable hardness limit is used to calculate the station-specific AWQC.

The aquatic benchmarks used to select COPCs in surface water in Section 4 are also AWQC values. In that instance, the chronic AWQC for both dissolved and total metals was compared the maximum detected concentration to identify a contaminant as a COPC. For the screening risk characterization, these comparisons are made for each surface water sampling station for both acute and chronic criteria. The results provide some insight on spatial trends of potential risks for aquatic life.

6.1.2 Screening Benchmarks for Sediment

Screening benchmarks for aquatic invertebrates for exposure to COPCs in sediment are identified based on a review of literature reporting sediment quality guidelines. Several sets of sediment quality guidelines are available. The National Oceanic and Atmospheric Administration (NOAA) compiled a set of Effects Range Low (ERL) and Effects Range Median (ERM) levels for contaminants in sediment (Long and Morgan, 1991). The Ontario Ministry of Environment has identified a set of Severe Effects Threshold (SET) values (Persaud et al., 1993). MacDonald et al. (1996) expanded on the work of Long and Morgan (1991) and developed a set of guidelines including threshold effects levels (TELs) and probable effects levels (PELs). These sediment quality guidelines are derived based on data primarily from marine environments.

Ingersoll et al. (1996) compiled freshwater sediment toxicity data from nine different sites in the United States and identified a series of sediment effect concentrations (SECs) for a series of metals in sediment. The SECs are defined as the concentrations of individual contaminants in sediment below which toxicity is rarely observed and above which toxicity is frequently observed. The database was compiled to classify toxicity data for Great Lakes sediment samples. Ingersoll et al.(1996) derived five different SECs according to the methodology of Long and Morgan (1990), Persaud et al. (1993) and MacDonald Environmental Sciences Ltd (1994). The SECs include an ERL, ERM, TEL, PEL and no effect concentration (NEC). Ingersoll et al (1996) calculated these freshwater ERL, ERM, TEL and PEL values using the same procedures as NOAA and MacDonald Environmental Sciences Ltd. (1994).

NOAA ERL and ERM Values. The NOAA ERL represents the 10th percentile of values sorted in ascending order reported to be associated with an adverse effect. The NOAA ERM is the median value in the ranking. An ERL is defined by Long and Morgan (1990) and Long et al. (1995) as the concentration of a chemical in sediment below which adverse effects are rarely observed or predicted among sensitive species. An ERM is defined by Ingersoll et al (1996) as the concentration of a chemical above, which effects are frequently or always observed or predicted among most species. The ERLs calculated by Ingersoll et al. (1996) use the 15th percentile.

State of Florida TEL and PEL Values. MacDonald Environmental Sciences Ltd .(1994) calculated TELs and PELs using an expanded database of Long and Morgan (1991). Freshwater data were excluded from the analyses. Sediment concentrations associated with an adverse effect were sorted in ascending order and an ERL (15th percentile) and ERM (50th percentile) were identified. The concentrations associated with no adverse effect were also sorted and a no effect range high (85th percentile) and no effect range median (50th percentile) were identified. The TEL is equal to the geometric mean of the ERL and no effect range median. The PEL is equal to the geometric mean of the ERM and the no effect range high. Although similar, the TEL and PEL values are lower than the ERL and ERM values. The values are lower because they are calculated using both "effect" and "no-effect" data; whereas, the ERL and ERM use only "effect" data. The NEC is the maximum concentration of a chemical in sediment that does not significantly adversely affect the particular response when compared to the control.

Consensus-Based Sediment Quality Guidelines (SQGs). In an effort to focus on agreement among the various sediment quality guidelines (previously discussed), MacDonald et al. (2000) issued consensus-based SQGs for 28 chemicals of concern. For each chemical of concern, a threshold effect concentration (TEC) and a probable effect concentration (PEC) were identified. The predictive reliability of these values was also evaluated. The criteria for establishing reliability of the consensus-based PECs was based on Long et al. (1998). This predictive ability analysis was focused on the ability of each SQG when applied alone to classify samples as either toxic or non-toxic. These criteria are intended to evaluate the narrative intent of the values. Sediment toxicity should be observed only rarely below the TEC and should be frequently observed above the PEC. Individual TECs were considered reliable if more than 75% of the sediment samples were correctly predicted to be non-toxic. Similarly, the individual PEC was considered reliable if greater than 75% of the sediment samples were correctly predicted to be toxic. Therefore the target levels of both false positives (samples incorrectly classified as toxic) and false negatives (samples incorrectly classified as non toxic) was 25% using the TEC and PEC. The SQGs were considered to be reliable only if a minimum of 20 samples were included in the predictive ability evaluation (MacDonald et al., 2000). The results of the reliability analyses is summarized in the following table:

Reliability of Individual Consensus-Based Sediment Quality Guidelines (MacDonald et al., 2000)

Chemical	% of Samples Correctly Predicted to Be Non-Toxic based on TEC	TEC Reliable?	% of Samples Correctly Predicted to Be Toxic based on PEC	PEC Reliable?
Arsenic	74.1%	No	76.9%	Yes
Cadmium	80.4	Yes	93.7	Yes
Chromium	72.0	No	91.7	Yes
Copper	82.3	Yes	91.7	Yes
Lead	81.6	Yes	89.6	Yes
Mercury	34.3	No	100	Yes
Nickel	72.3	No	90.6	Yes
Zinc	81.6	Yes	90.0	Yes

Because field collected sediments contain a mixture of chemicals, a second predictive analyses was completed for use of the individual SQGs together in classifying a sediment as toxic or non-toxic. The incidence of effects was noted above and below various mean PEC quotients (ratios). The mean PEC ratio equals the average of the ratios of the concentration of the chemical to the corresponding PEC using on the PEC values that were found to be reliable. 92% of sediment samples with a mean PEC quotient > 1.0 were toxic to one or more species of aquatic organisms. The relationship between PEC quotient and incidence of toxicity is depicted in Figure 6-1. The mean PEC quotient was found to be highly correlated with incidence of toxicity ($r^2 = 0.98$) (MacDonald et al., 2000).

For the SERA, consensus-based SQGs from MacDonald et al. (2000) are used as a range of toxicity benchmarks for sediment. The TEC is used as the low benchmark and the PEC as the high benchmark. Consensus values are not available for aluminum, antimony, barium, beryllium, cobalt, cyanide, manganese, selenium, silver, thallium or vanadium. For aluminum and manganese the lowest and highest SEC values from Ingersoll et al. (1996) are used as the range of toxicity benchmarks for sediments. For silver, sediment toxicity benchmarks are the range of values reported by NOAA (ERL and ERM) (Long et al., 1995) and the state of Florida (MacDonald Environmental Sciences Ltd., 1994). For antimony the benchmarks are the range of values reported by Long and Morgan (1991). Sediment toxicity benchmarks could not be identified for barium, beryllium, cobalt, cyanide, selenium, thallium and vanadium.

For the SERA, the identified low and high sediment toxicity benchmarks are listed in Table 6-2. These values are compared to the EPC values for sediments for each sampling location (Section 5.1.2) to evaluate risks for benthic invertebrates for direct contact with COPCs in sediment in Section 7.1.2.

6.2 Toxicity Benchmarks for Amphibians

Screening benchmarks for the protection of amphibians from aqueous direct contact exposures are identified for several endpoints from the EPA AQUIRE database. With the exception of cyanide, the data available are LC50 values which represent a test concentration lethal to 50% of the test population. To estimate a toxicity benchmark value for no adverse effects, the lowest LC50 from the database is selected and the concentration is divided by ten. The only available endpoint for cyanide is avoidance behavior. Selected benchmarks are presented in Table 6-3. It should be noted that these benchmarks serve as screening values that do not account for site-specific factors which may either increase or reduce toxicity.

The toxicity screening benchmark for each COPC is compared to the EPC value for surface water and seep water to calculate HQ values in Sections 7.2.1 and 7.2.2, respectively.

6.3 Plant Toxicity Benchmarks

6.3.1 Screening Benchmarks for Soil

Plants are exposed to metals in soil principally through their roots. Exposure may also occur due to deposition of dust on foliar (leaf) surfaces, but this pathway is believed to be small compared to root exposure. Copper and zinc are considered to be essential or beneficial for plant growth (Kabata-Pendias and Pendias, 1992). However, excessive levels of these and other metals in soil may exert a variety of adverse effects on plants including reduced photosynthetic efficiency, reduced seed germination, and reduced root-mass formation. These phytotoxic responses may occur at the scale of the individual plant or may effect the entire plant community, resulting in areas of stressed and unhealthy vegetation. Stressed communities are often subject to invasion by weedy metals-tolerant species which in turn can result in the disruption and displacement of an entire plant community that would otherwise be found in an affected area. In some locations, lethality to plants can result, and areas with little or no vegetative cover may occur.

A relatively large body of literature exists regarding metal phytotoxicity. These studies show that the toxicity of metals in soils varies widely between different plant species, and also depends on a large number of soil parameters including soil type, organic content, water content, soil condition, soil chemistry, and soil pH (Adriano, 1986; Kabata-Pendias and Pendias, 1992; CH2MHill, 1987a; CH2MHill, 1987b; Efroymson et al., 1997a). This variability is evident by inspection of Table 6-4, which summarizes phytotoxicity benchmarks for metals that are recommended and used by different authors and groups.

These values vary over an order of magnitude or more for each metal. Screening benchmarks for cyanide and selenium could not be identified.

The low and high toxicity values identified in Table 6-4 are compared to EPCs in soil for each sampling location to evaluate risks for terrestrial plants in Section 7.3.1.

6.3.2 Screening Benchmarks for Water

Screening benchmarks for the protection of plants from aqueous exposures are available from the Oak Ridge National Laboratory (ORNL) (Efroymson et al., 1997a). The screening benchmarks developed by ORNL are assumed to be representative of exposures of plants to contaminants measured in soil solutions (e.g., from lysimeter samples or possibly from aqueous extracts of soil) or in very shallow groundwater (e.g., plants in the vicinity of seeps and springs).

Solution benchmarks include data from toxicity tests conducted using whole plants rooted in aqueous solutions. Tests are commonly conducted in this manner because plants are assumed to be exposed to contaminants in the solution phase of soil, and the presence of soil in test systems reduces the experimenter's degree of control over exposure (Efroymson et al., 1997a). It should be noted that these benchmarks are used for screening and do not account for site-specific soil and plant characteristics.

The phytotoxicity benchmarks are derived by rank-ordering the LOEC values and then selecting a benchmark that approximated the 10th percentile. If there were 10 or fewer values for a chemical, the lowest LOEC is used. If there are more than 10 values, the 10th percentile LOEC value is used. If the 10th percentile fell between LOEC values, a value is chosen by interpolation. Since these benchmarks are intended to be thresholds for significant effects on growth and production, test endpoints that indicate a high frequency of lethality are not appropriate. Therefore, when a benchmark is based on an LC50 or on some other endpoint that includes a 50% or greater reduction in survivorship, the value is divided by a factor of 5, an approximation of the ratio of the LC50 to the EC20. In all cases, benchmark values are rounded to one significant figure. The selected toxicity benchmarks for plants for aqueous exposures are presented in Table 6-5. These benchmarks are compared to EPCs for seep water (Section 5.2.2) in Section 7.3.2 to evaluate risks for terrestrial plants associated with exposure to COPCs in seep water (groundwater data).

6.4 Soil Fauna Toxicity Benchmarks

Soil organisms are defined as organisms that live during an essential part of their life cycle in the soil. This includes both soil invertebrates (e.g., worms, some insects and arthropods, etc), and soil microbes (bacteria, fungi, etc.). Soil organisms are important components of the terrestrial ecosystem as prey for other species, and because they contribute substantially to litter breakdown. Soil invertebrates fragment and partially solubilize organic matter, while soil microorganisms mineralize complex organic molecules to simple molecules that can be taken up by roots, or further mineralized to CO_2 and H_20 (Eijsackers, 1994). Earthworms are probably the most important soil invertebrate in promoting soil fertility (Edwards, 1992). Their feeding and burrowing activities break down organic matter and release nutrients and improve aeration, drainage and aggregation of soil.

Soil organisms are distinguished as inhabitants of either pore water, mineral soil or the litter layer. Some scientists distinguish between "in soil" and surface-active organisms, but this distinction can be arbitrary and is not considered for this assessment. Soil organisms can be exposed to contaminants in soils by direct contact with metals in pore water, and ingestion of metals in mineral soil or the litter layer. Site-specific soil and invertebrate characteristics can influence the bioavailability and resulting toxicity of metals from the soils to soil organisms (Eijsackers, 1994).

Soil screening benchmarks for the protection of soil organisms and microbial processes are available from three different sources, including ORNL (Efroymson et al., 1997b), the National Institute of Public Health and the Environment (Bilthoven, the Netherlands) (RIVM, 1997), and the Canadian Council of Ministries of the Environment (CCME, 1997).

The screening benchmarks developed by ORNL for application at hazardous waste sites (Efroymson et al., 1997b) are derived using a method similar to that used by NOAA to establish the ERLs and ERMs for sediment (Long and Morgan, 1990). The data available on toxicity of a contaminant to soil organisms were reviewed and the lowest observed effect concentration (LOEC) was determined. The LOEC is defined as the lowest applied concentration of the chemical causing a greater than 20% reduction in the measured response. In some cases, the LOEC was the lowest concentration tested or the only concentration reported (EC50 or ED50 data). The LOECs were rank ordered and a value selected that approximated the 10th percentile. When a benchmark was based on a lethality endpoint, the benchmark value was divided by 5 to approximate an effects concentration for growth and reproduction. The factor was selected based on the author's judgement. The benchmark values were then rounded to one significant figure (Efroymson et al., 1997b). Efroymson et al. (1997b) developed screening benchmarks for earthworms and microorganisms and microbial soil processes.

The values developed by each of these groups are summarized in Table 6-6. As seen, in most cases the benchmarks developed by the different groups for each chemical vary by less than an order of magnitude. An exception is mercury, for which the range of soil invertebrate TRVs is substantially wider (300-fold). Screening benchmarks for antimony and cyanide could not be identified.

For the purposes of the SERA, the low and high toxicity benchmarks are compared to soil EPCs for each sampling location (Section 5.2.1) to calculate a range of HQ values in Section 7.4.

6.5 Wildlife Toxicity Reference Values (TRVs)

Two toxicity reference values (TRVs) are identified for each COPC for each representative wildlife species. The first TRV is an estimate of the dose (mg of contaminant per kg of body weight per day) that is not associated with any adverse effects to the species. This is referred to as the no observed adverse effect level (NOAEL) TRV. The second TRV is an estimation of the dose that first causes an observable adverse effect, and is referred to as the lowest observed adverse effect level (LOAEL) TRV. This range of TRVs is one way to bracket the true threshold for adverse effects.

The NOAEL and LOAEL TRVs are based on a critical review of published toxicity data. Two secondary sources (Sample et al., 1996 and Engineering Field Activity West, 1998) were used to identify key toxicological studies for each of the COPCs. The studies were reviewed to determine the relevance and reliability of the study results for derivation of a TRV. The critical studies used to derive the TRVs are presented in detail for each contaminant and each receptor in Appendix D.

Separate TRVs (both NOAEL-based and LOAEL-based) were developed for exposure via water and the diet. This distinction is based on the observation that the absorption (and hence the toxicity) of metals in the diet is usually lower than for metals dissolved in water. Both the water TRVs and the dietary TRVs were based on published toxicity data, wherever possible. If toxicity data were available for only one of these media (water or diet, but not both), a relative absorption factor of 50% was assumed to extrapolate to the other medium:

$$TRV(water) = TRV(diet) \times 0.50$$
$$TRV(diet) = TRV(water) / 0.50$$

This adjustment factor of 50% is based on professional judgement, but is supported by evidence that metals in water typically exist in a readily bioavailable form, and that dietary materials (proteins, carbohydrates, other minerals) tend to bind metals and/or compete for uptake sites, hence reducing their bioavailability. This concept has been used previously by USEPA in the derivation of food- and water-based Reference Doses (RfDs) for cadmium (IRIS, 1998).

In theory, separate TRVs are needed for sediment and soil ingestion, since absorption of contaminants from sediment may not be the same as from either food or water. However, there are no toxicity data for any of the COPCs to any of the representative wildlife species where the exposure occurs in the form of soil. Therefore, TRVs for food were used as surrogates for sediment and soil TRVs. It is considered likely that this approach may tend to overestimate exposure and risk from ingestion of sediment and soil, but this is not known for certain.

When reliable toxicity data could not be located for a representative species, it was necessary to extrapolate toxicity data from studies using another species. In some cases, available toxicity data were too limited to allow precise definition of NOAEL and LOAEL values for relevant endpoints. To account for these data gaps, each TRV was derived from the study dose level identified as the NOAEL or LOAEL by dividing by an Uncertainty Factor (UF) as follows:

$$TRV = Study\ Dose / UF$$

The value of UF was calculated as the product of a series of sub-factors. These sub-factors of uncertainty are presented in Table 6-7 and include inter-taxon extrapolation, exposure duration, toxicological endpoint, and other modifying factors such as threatened and endangered status, contaminant sensitivity, developmental differences, etc. In general, USEPA Region VIII recommends that HQ values be calculated only in cases where the total UF used to derive a TRV is less than 100. As seen in Appendix D, UFs used to derive TRVs are all below 100. The TRVs derived for each representative wildlife species are summarized in Table 6-8. The TRVs are compared to doses estimated for each wildlife species as described in Section 5.2 to estimate risks in Section 7.5.

7.0 SCREENING LEVEL RISK CHARACTERIZATION

Potential risks to ecological receptors from exposure to COPCs are characterized by use of a Hazard Quotient (HQ) approach. The HQ is defined as the ratio of the exposure point concentration (EPC) to the appropriate toxicity screening benchmark:

$$HQ = \frac{Exposure}{Benchmark}$$

If the effects of different chemicals on a receptor act on the same target tissue by the same mechanism, then the total Hazard Index (HI) to the receptor may be estimated as the sum of the chemical-specific HQ values across chemicals. At the RFT Site, it has been conservatively assumed that effects of all the metals on each of the receptors are additive.

$$Total\ HI_i \sum HQ_{i,r}$$

If the HQ or total HI is less than or equal to one, it is believed that unacceptable risks will not occur in the exposed population. If the HQ or total HI exceeds one, then unacceptable risks may occur and there is a need for further evaluation. All HQ and total HI values are presented to one significant digit.

7.1 Aquatic Receptors

7.1.1 Surface Water

Because the toxicity of COPCs in surface water to aquatic receptors is dependant on the length of exposure time, the HQ is calculated for both short-term (acute) and long-term (chronic) exposure conditions:

$$HQ_{acute} = \frac{C_{water}}{Benchmark_{acute}} \qquad HQ_{chronic} = \frac{C_{water}}{Benchmark_{chronic}}$$

The concentration of a contaminant in surface water may be expressed in terms of total recoverable metal or dissolved metal with the value of the denominator (benchmark) dependant on the type of concentration value selected.

$$HQ_{dissolved} = \frac{DissolvedC_{water}}{Benchmark_{dissolved}} \qquad HQ_{total} = \frac{TotalC_{water}}{Benchmark_{total}}$$

As discussed previously, the HQ based on the dissolved metal concentration is generally believed to be the best indicator of potential risks due to direct contact (e.g., gill respiration in fish), as this concentration represents the amount of the metal that is biologically available (USEPA, 1995). However, evaluation of risks to receptors based only on dissolved metal levels is not possible as dissolved benchmarks (criteria) are not available for all metals and dissolved measurements in surface water are not available for all COPCs for each surface water sampling station.

HQ values are calculated for COPCs in surface water and are presented in Table 7-1. The left-hand side of the table presents the total recoverable and dissolved COPC EPCs from each surface water sampling station. The corresponding acute and chronic AWQC values are also calculated. For those AWQCs that are dependant upon hardness, the average station hardness is used to derive the criteria. If the measured station hardness is outside of the specified hardness limits (Table 6-1), the applicable hardness limit are used to calculate the AWQC. If the station hardness is not available, a hardness of 200 mg/L is assumed. The right side of the table presents the resulting HQ_{acute} and $HQ_{chronic}$ values for dissolved and total recoverable COPCs. Where the HQ values exceed 1E+00, the values are in boldface type.

Figure 7-1 provides a plot of HQ values for all COPCs by surface water station. The lower point of the plotted range represents the HQ value calculated using the acute AWQC and the higher point represents the HQ value calculated using the chronic AWQC. Acute and chronic AWQC values for zinc are nearly equal depending on hardness, therefore a range of HQs is not presented for all stations.

Each of the following subsections discusses the surface water HQ results for both total recoverable and dissolved measurements for each COPC in which an exceedance of either acute or chronic toxicity screening levels (AWQC) occurs.

- ***Upstream Silver Creek.*** Zinc concentrations (both total and dissolved) at all sampling locations on Silver Creek upstream of the railroad bridge trestle are above levels associated with acute and chronic toxicity for aquatic receptors. At these stations, exceedances of the chronic toxicity criteria for total and dissolved cadmium are also observed with total cadmium levels also exceed the acute toxicity levels at station 492695. Total lead concentrations are above a chronic level of concern at all sampling locations with HQs ranging from 3 to 3,000. At sampling location N4, total concentrations of copper and mercury are above levels of acute and chronic toxicity. The dissolved concentrations of lead at station N4 are also above a chronic level of concern (HQ of 5) with total concentrations above an acute level of toxicity. Immediately upstream of the railroad bridge trestle (USC-3), dissolved aluminum concentrations are slightly above chronic toxicity levels (HQ of 2). At the furthest upstream location (USC-7), below Silver Maple Claims, total aluminum concentrations are also above chronic levels (HQ of 8).

- ***Downstream Silver Creek.*** Like the upstream section of Silver Creek, zinc concentrations (both total and dissolved) at all but three sampling locations on Silver Creek downstream of the railroad bridge trestle are above levels associated with acute and chronic toxicity for aquatic receptors. At three locations (RF-SW-06, USC-1 and RF-8) total aluminum concentrations are above chronic toxicity levels. Total and dissolved concentrations of cadmium are above chronic toxicity levels at all sampling locations except station 492679.. At most sampling locations, total lead concentrations (and often dissolved concentrations) are above a level of chronic toxicity. Total mercury concentrations at station N6 are above acute and chronic toxicity levels (HQs range from 90 to 200).

- ***South Diversion Ditch.*** At most sampling locations in the south diversion ditch, both total and dissolved zinc concentrations are above levels associated with acute and chronic toxicity. Total zinc concentrations at RF-4 and RF-5-4 are 10 times greater than chronic toxicity levels. Dissolved chromium concentrations are above levels associated with acute toxicity at stations USC-4 and RF-6. Total concentrations of chromium are 7 times greater than chronic toxicity levels and 4 times greater than acute toxicity levels at USC-4. Total aluminum concentrations are above levels associated with chronic toxicity at most sampling locations with dissolved aluminum concentrations above a level of chronic toxicity at station RF-2. At RF-6-2, total arsenic concentrations exceed acute and chronic toxicity levels. Total lead concentrations slightly exceed levels of chronic toxicity (HQs ranging from 2 to 9) at several stations.

- ***Ponded Water on the Impoundment.*** The HQs for each COPC are below levels of acute and chronic toxicity. However, the total HI is above one for both total and dissolved metals based on chronic toxicity criteria and above one for dissolved metals based on acute toxicity criteria.

- ***Unnamed Drainage flowing into the South Diversion Ditch.*** At sampling location RF-3-2, all total and dissolved COPC concentrations, with the exception of total recoverable aluminum, are below levels of acute and chronic toxicity. Total aluminum concentrations are above levels of acute and chronic toxicity levels (HQs of 2 and 20, respectively).

The range of HQ values for aquatic receptors from surface water are summarized below.

Location	Al	As	Cd	Cr	Cu	CN	Pb	Hg	Se	Ag	Zn
Silver Creek - upstream	<1 to 8	All <1	<1 to 30	All <1	<1 to 20	All <1	<1 to 300	<1 to 200	<1to 2	All <1	3 to 400
Silver Creek - downstream	<1 to 4	All <1	<1 to 20	All <1	<1to 2	All <1	<1 to 60	<1 to 200	<1to 2	All <1	<1 to 8
South Diversion Ditch	<1 to 7	<1 to 5	<1 to 4	<1 to 7	All <1	All <1	<1 to 9	All <1	All <1	All <1	<1 to 10
Ponded Water	NA	All <1	<1to 2	All <1	All <1	NA	All <1	All <1	All <1	All <1	All <1
Unnamed Drainage	<1 to 20	All <1	All <1	All <1	All <1	NA	All <1	All <1	All <1	All <1	All <1

The concentrations of most COPCs are above levels of chronic and/or acute toxicity in Silver Creek upstream of the RFT Site. The headwaters of Silver Creek originate in the mountains south of Park City, Utah and include Deer Valley, Empire Canyon, Ontario Canyon, and Thaynes Canyon (Figure 3-6). Historically, these headwaters were the site of several mining operations such as the Little Bell and Daly

Mines. According to the Utah Division of Water Quality, water quality in the upstream portions of Silver Creek is impaired and concentrations exceed the state water quality standards for zinc (RMC, 2000b). During the watershed evaluation completed by EPA (USEPA, 2001a), surface water samples were collected at several locations in each canyon and along Silver Creek (see Figure 3-7). Measured surface water concentrations of cadmium, lead and zinc are presented graphically in Figure 7-2.

As seen in Figure 7-2, the highest concentrations of cadmium, lead and zinc are measured in Empire Canyon. Concentrations in Silver Creek tend to decrease with increasing distance downstream with increases observed at locations near Silver Maple Claims that receives flow from the Pace-Homer Ditch. According to the findings of the watershed evaluation (USEPA, 2001a), the Silver Maple Claims (Pace-Homer Ditch) was the largest contributor of zinc for the lower reaches of Silver Creek. Zinc loads from the RFT Site south diversion ditch are reported to contribute only 0.03 lbs/day to Silver Creek (USEPA, 2001a).

The following subsections provide further evaluation of the risks for cadmium, lead and zinc in surface water for fish and aquatic invertebrates, respectively.

7.1.1.1 Screening Evaluation for Fish

The "typical" concentrations of cadmium, lead and zinc in RFT Site surface waters are compared to species specific toxicity reference values (species mean TRVs). Figures 7-3a to 7-3c compare data on the available mean and maximum concentrations of dissolved cadmium, lead and zinc observed in Silver Creek and RFT Site surface waters to the range of species-mean toxicity values for the fish species that either occur in or are similar to species that occur in cold water streams (Table 7-2). The data for the south diversion ditch and the unnamed drainage is provided for comparison purposes. It is understood that this habitat is semi-permanent and is not expected to support a cold water fishery.

All of the toxicity values shown in Table 7-2 are derived from the corresponding AWQC Documents prepared by EPA (1985b-e, 1987, 1996, 2001b). Because the toxicity of cadmium, lead and zinc depend on water hardness, all of the data (both the toxicity values and the concentration values) are normalized to a default hardness of 100 mg/L using the following equation:

$$C(100) = C(H) \times TRV(100) / TRV(H)$$

where:

C(100) = normalized concentration
C(H) = original concentration (hardness = H)
TRV(100) = Acute AWQC (dissolved) at a hardness of 100 mg/L
TRV(H) = Acute AWQC (dissolved) at hardness = H

Site-specific data on water hardness are not available for all stations. If the station hardness is not available, a hardness of 200 mg/L is assumed.

For dissolved cadmium (Figure 7-3a), average concentrations for several locations in Silver Creek and maximum cadmium concentrations in the south diversion ditch enter a range of acute toxicity for brook trout and rainbow trout. As seen in Figure 7-3b, dissolved lead concentrations do not enter a range of acute or chronic toxicity for either brook trout or rainbow trout at any location, even when concentration values reach the maximum detected concentrations. For zinc (Figure 7-3c), average concentration values at station RF-7 in upstream Silver Creek exceed acute and chronic toxicity values for all fish species. All other zinc concentrations are below available species toxicity values.

7.1.1.2 Screening Evaluation for Aquatic Invertebrates

Many benthic macroinvertebrates live some or most of their life cycle on or near the surface of the sediment substrate, and hence the main source of water exposure is from the overlying surface water column (Warren et al., 1998). Data on the concentration of metals in surface water are presented earlier (see Section 3). In accord with EPA recommendations (Prothro, 1993), attention is focused on risks from contact with dissolved metals, since dissolved metal measurements are thought to be more predictive of risk compared to measurements of total recoverable metals.

Table 7-3 summarizes available water column toxicity data from the AWQC national database (USEPA, 1985b-e, 1987, 1996, 2001b) for benthic species that are expected to occur or are reasonable surrogates for other species that are expected to occur in the RFT Site waters. Daphnia are retained because they are usually among the most sensitive of aquatic invertebrates to the effects of metals, and therefore can serve as a surrogate for other sensitive aquatic macroinvertebrates which may reside in RFT Site surface waters, but standard toxicity values are not available.

Figures 7-4a to 7-4c compare data on the distribution of concentrations of dissolved metals observed in RFT Site surface waters to the range of genus-mean toxicity values for aquatic macroinvertebrates selected to represent the aquatic macroinvertebrate community. Because cadmium, lead and zinc toxicity depends on water hardness, all of the data (both the toxicity values and the concentration values) have been normalized to a hardness of 100 mg/L. The hardness-normalization equation is presented previously in Section 7.1.1.1. Site-specific data on water hardness are not available for all stations. If the station hardness is not available, a hardness of 200 mg/L is assumed.

For dissolved cadmium (Figure 7-4a), concentrations approach or exceed chronic toxicity values for cladocerans (Daphnia) at several locations in Silver Creek and the south diversion ditch. As seen in Figure 7-4b, dissolved lead concentrations do not enter a range of acute or chronic toxicity for any benthic macroinvertebrate genus or species evaluated at any location, even when concentration values reach the maximum detected concentrations. However for zinc (Figure 7-4c), average concentrations in Silver Creek and the south diversion ditch are frequently above levels of chronic toxicity for cladocerans (Daphnia). In addition, maximum concentration values in the south diversion ditch (RF-4 and RF-5-4) approach or exceed reported acute toxicity levels for Daphnia. These comparisons suggest that these and other aquatic invertebrate organisms may be exposed to cadmium and zinc concentrations that could impact or limit their populations.

7.1.2 Sediments

Risks for benthic invertebrates from exposures to COPCs in sediment are evaluated using two methods. The first is a HQ approach and the second is calculation of site-specific probable effect ratios that predict if the mixture of metals in site sediments will be toxic to benthic organisms.

7.1.2.1 Hazard Quotients

The risks to benthic invertebrates from exposures to COPCs in sediment are evaluated using an HQ approach as follows:

$$HQ = \frac{C_{sed}}{Benchmark_{sed}}$$

where:

C_{sed}	=	Concentration of COPC in sediment (mg/kg dry weight)
$Benchmark_{sed}$	=	Sediment screening benchmark (mg/kg dry weight)

Table 7-4 presents the maximum concentration of each COPC in sediment, stratified by location, with the corresponding range of sediment screening benchmarks (low and high toxicity benchmarks). HQs are calculated using both the low and high sediment toxicity benchmarks. The resulting range of HQ values are shown on the right-hand side of Table 7-4. In instances where the HQ exceeds 1, the HQ is shown in boldface type.

Figure 7-5 presents a plot of HQ values for aluminum, arsenic, cadmium, copper, lead, manganese, mercury, nickel, silver, and zinc stratified by sediment station. The lower point on the range represents the HQ value calculated using the high sediment toxicity benchmark (Table 6-2) and the higher point represents the HQ value calculated using the low sediment toxicity benchmark.

Based on the HQ values, potential risks for benthic invertebrates are predicted for exposures to aluminum, antimony, arsenic, cadmium, chromium, copper, lead, manganese, mercury, nickel, silver and zinc in sediments. The HQ values for cadmium, lead and zinc tend to follow similar trends across locations. A discussion of the HQ values for benthic invertebrates are provided by COPC in the following subsections:

- ***Upstream Silver Creek***. Antimony, arsenic, cadmium, copper, lead, mercury, silver and zinc sediment concentrations at all sampling locations along Silver Creek upstream of the railroad trestle are above levels associated with sediment toxicity to benthic invertebrates. Antimony, arsenic, cadmium, copper, lead, silver and zinc concentrations exceed both the low and high sediment toxicity benchmarks at all three upstream sampling locations. Mercury sediment concentrations exceed the low toxicity benchmark at all three upstream sampling locations but only station USC-6 exceeds the high toxicity benchmark (HQ of 2). Aluminum and chromium sediment concentrations are below a level of concern for benthic invertebrates (HQs less than or equal to 1) at all three upstream Silver Creek locations. The highest HQs are for COPCs observed at the sampling station below Silver Maple Claims (USC-6). At this station, the risks are predicted to range from 9 (mercury) to 1,000 (lead) based on the low toxicity benchmark values (TECs), and from 2 (mercury) to 300 (lead) based on the high toxicity benchmark values (PECs).

- ***Downstream Silver Creek.*** The HQ values for benthic invertebrates from direct contact with sediment at Silver Creek sampling locations downstream of the RFT Site are similar to those at upstream locations. At both Silver Creek downstream sampling locations antimony, arsenic, cadmium, copper, lead, silver and zinc sediment concentrations are higher than both the low and high toxicity benchmarks. At station USC-1, concentrations of mercury are higher than the low toxicity benchmark (HQ of 2). Aluminum and chromium concentrations are less than both benchmarks (HQs less than or equal to 1).

- ***South Diversion Ditch.*** Antimony, arsenic, cadmium, lead, silver and zinc sediment concentrations at almost all sampling locations in the south diversion ditch are above both the low and high toxicity benchmarks. Concentrations of copper exceed both the high and low toxicity benchmarks with the exception of locations RF-SD-SD2, -SD3, and -SD6. The concentrations of mercury in sediments exceed the low sediment toxicity benchmark at all sampling locations and the high benchmark at one location (RF-SD-SD1). Concentrations of aluminum and chromium are lower than both benchmarks with the exception aluminum at one station (RF-SD-SD6) where the HQ is 2. The highest HQ values are observed for cadmium, lead and zinc with values ranging from 20 (cadmium) to 100 (lead and zinc). The HQ ranges for other COPCs are generally lower.

- ***Wetland Area.*** Antimony, arsenic, cadmium, lead, silver and zinc exceed both the high and low toxicity benchmarks at all sampling locations. Concentrations of copper exceed the low toxicity benchmark at all locations and the high benchmark at two sampling locations (RF-SE-01 and RF-SE-03). The concentrations of mercury exceed the low toxicity benchmark at three locations and only the high benchmark at one location. Concentrations of manganese exceed the low toxicity benchmark at all locations and the high benchmark at all but one location (HQs range from 2 to 50). Concentrations of nickel exceed the low toxicity benchmark at RF-SE-01 and both the low and high toxicity benchmarks at RF-SE-04. Concentrations of aluminum and chromium are below a level of concern at all sampling locations.

The range of HQs and the relative frequency of exceedances for benthic invertebrate receptors from sediments are summarized in the following table.

Location	Al	Sb	As	Cd	Cr	Cu	Pb	Mn	Hg	Ni	Ag	Zn
Silver Creek - upstream	All <1	2 to 300	2 to 100	5 to 100	All <1	3 to 60	10 to 800	*NC*	<1 to 5	NC	7 to 100	8 to 300
Silver Creek - downstream	All <1	5 to 70	7 to 30	8 to 50	All <1	3 to 20	50 to 200	*NC*	<1 to 2	NC	10 to 50	20 to 80
South Diversion Ditch	<1 to 2	<1 to 50	3 to 20	4 to 70	All <1	<1 to 10	20 to 100	*NC*	<1 to 9	NC	4 to 30	6 to 100
South Diversion Ditch - Wetland	<1 to 2	2 to 50	5 to 30	8 to 90	All <1	<1 to 20	20 to 200	<1 to 70	<1 to 40	<1 to 4	2 to 50	10 to 100

NC= Not Calculated

As seen, sediments in upstream Silver Creek (above the RFT Site) tend to have the highest HQ values. According to the watershed evaluation (USEPA, 2001a), sediment concentrations are highest at and below Silver Maple Claims and are likely impacted by the tailings piles along the lower portions of Silver Creek. Historical releases from the RFT Site south diversion ditch may have also impacted sediments in Silver Creek (USEPA, 2001a).

7.1.2.2 Mean Probable Effect Concentration Ratio

As described earlier in Section 6, MacDonald et al. (2000) found that the mean PEC quotient was correlated with incidence of sediment toxicity ($r^2 = 0.98$). The resulting equation ($Y=101.48(1-0.36^x)$), where 'x' equals the mean PEC quotient and 'Y' equals the incidence of toxicity, can be used to estimate the probability of observing sediment toxicity at any mean PEC quotient. The mean PEC quotients calculated for each sediment sampling location are provided in Table 7-5 and the results are summarized in the following text table:

Calculation of the Mean PEC Quotient by Sampling Location and the Predicted Incidence of Observing Sediment Toxicity (MacDonald et al., 2000)			
Location	**Station**	**Mean PEC**	**Probability of Observing Sediment Toxicity**
Silver Creek Downstream	USC-1	19.8	100%
	USC-2	14.9	100%
Silver Creek Upstream	USC-5	21.3	100%
	USC-6	77.2	100%
	USC-7	6.5	100%
South Diversion Ditch	RF-SD-SD1	10.9	100%
	RF-SD-SD2	7.6	100%
	RF-SD-SD3	6.0	100%
	RF-SD-SD4	8.8	100%
	RF-SD-SD5	7.4	100%
	RF-SD-SD6	4.9	100%
Wetland Area	RF-SE-01	17.4	100%
	RF-SE-02	8.8	100%
	RF-SE-03	13.2	100%
	RF-SE-04	6.7	100%

The mean PEC ratio equals the average of the individual COPC specific ratios of the concentration of the COPC in sediment to the corresponding PEC value using only the PEC values that were found to be reliable. The mean PEC quotients for all sampling locations predict that samples are toxic to benthic invertebrates.

7.1.3 Seep Water

Potential risks for aquatic receptors from exposure to COPCs in seep water are characterized by use of the HQ approach. The HQ is defined as the ratio of the concentration of a COPC to the appropriate benchmark value:

$$HQ = \frac{C_{seep}}{Benchmark_{seep}}$$

where:

C_{seep} = Dissolved or Total Concentration of COPC in seep water (ug/L)
$Benchmark_{seep}$ = AWQC screening benchmark for Total or Dissolved Concentrations (ug/L)

HQ values for aquatic receptors are calculated for COPCs in seep water (as estimated from groundwater) and are presented in Table 7-6. The left-hand side of the table presents the maximum total recoverable and dissolved COPC concentrations from each groundwater monitoring well. The corresponding acute and chronic AWQC values are also presented. Where the HQ values exceed 1E+00, the values are in boldface type. Calculated HQs for total and dissolved COPCs are shown graphically in Figure 7-6. If the value of the HQ exceeds one, then potential risks may occur and there is a need for further evaluation.

A summary of the seep water HQ results for each COPC in which an AWQC exceedance occurred is provided below.

Location	Al	As	Cd	Cr	Cu	CN	Pb	Hg	Se	Ag	Zn
Seep Water @ Main Embankment	<1 to 900	<1 to 2	<1 to 90	<1 to 9	<1 to 90	<1 to 2,000	<1 to 30	<1 to 3	<1 to 3	All <1	<1 to 10
Background Groundwater	<1 to 200	All <1	<1 to 8	<1 to 2	<1 to 10	All <1	<1 to 100	All <1	All <1	All <1	All <1

The ranges of HQ values exceed one for all COPCs, with the exception of silver, at all monitoring wells located at the base of the main embankment. Total concentrations have consistently higher HQ values than those predicted for dissolved. Concentrations of cyanide along with lead and mercury are found to be the most common contributors to risks.

7.2 Amphibians

The diversity, density, and the reproductive success (i.e. embryonic mortality) of amphibians are shown to be sensitive indicators of environmental stress. If amphibians are found to encounter reproductive failure compared to reference wetlands, amphibian reproductive success and diversity, and subsequently structure and function as a whole would be determined to be at risk.

The basic equation used for calculation of an HQ value for the direct contact exposure of amphibians to COPCs in aqueous media is:

$$HQ_{amphib} = \frac{Conc_{water}}{TB_{amphibian}}$$

where:

$Conc_{water}$ = Total Recoverable concentration of COPC in water (ug/L)
$TB_{amphibian}$ = Toxicity benchmark (ug/L) for exposure of amphibians to COPCs in aqueous media

HQ values are calculated using the amphibian toxicity benchmark TRV for each COPC. If all HQ values are found to be below one, it would then be concluded that hazard to amphibians from exposure to COPCs in water is low. Conversely, if a majority of HQ values based on the benchmark TRV are found to be substantially higher than one, it should be concluded that toxicity to amphibians from exposure to COPCs in water is likely.

7.2.1 Surface Water

HQ values for the exposure of amphibians via surface water are calculated for each COPC and are presented in Table 7-7. The left-hand side of the table presents the maximum total recoverable COPC concentrations from each surface water sampling station. If total concentrations are not available, the dissolved concentrations are used to calculate HQs. The corresponding amphibian toxicity benchmark screening values are also presented. Where the HQ values exceed 1E+00, the values are in boldface type. A summary of the total HI at each sampling station and the contribution of each COPC HQ to the total HI is presented in Figure 7-7.

A summary of the surface water HQ results for each COPC in which an exceedance of the amphibian toxicity screening benchmark occurs is provided in the following paragraphs.

- ***Upstream Silver Creek.*** Zinc and copper concentrations at all sampling locations and lead concentrations at all but one sampling location are above levels associated with toxicity to amphibians. Copper HQs typically are less than 5 times greater than the toxicity value. Slight exceedances of the cadmium and arsenic toxicity benchmarks are observed at several sampling locations with maximum HQs of 3 and 5, respectively. Cyanide concentrations at sampling location RF-7-2 and N4 are also above the toxicity value, with HQs of 8 and 200, respectively. Mercury concentrations at these stations and at station RF-7 exceed of the toxicity value as well. Selenium and silver concentrations are below respective toxicity values at all stations.

- ***Downstream Silver Creek.*** The HQ values and frequency of exceedances of amphibian toxicity values at locations in Silver Creek downstream of the south diversion ditch confluence are similar to those observed upstream. Like upstream Silver Creek, zinc and lead concentrations at all but one sampling location are above respective toxicity values. Arsenic and copper HQs are greater than 1 at all but one location, with maximum HQs of 8 and 3, respectively. At station RF-8, cadmium concentrations are slightly above the toxicity value (HQ of 2). Cyanide is measured at only three sampling locations, but concentrations are above the toxicity value at all locations with a maximum HQ of 20. Calculated HQs for mercury at most locations are below 1, however, HQs are greater than 1 at N6, RF-8, and RF-8-2. Similar to upstream Silver Creek, selenium and silver concentrations are below toxicity values at all stations.

- ***South Diversion Ditch.*** Total zinc concentrations at every sampling location in the south diversion ditch are above toxicity levels. Zinc concentrations at RF-4 and RF-5-4 exceed the toxicity value by 3,000 times. At most sampling locations, total arsenic concentrations (HQs ranging from <1 to 200), total copper concentrations (HQs ranging from <1 to 5) and total mercury concentrations (HQs ranging from <1 to 3) exceed respective toxicity values. Concentrations of lead at several locations in the south diversion ditch are also above the toxicity value with a maximum HQ of 10. Total cyanide is available for only one location. At this location concentrations are 8 times greater than the toxicity value. Cadmium, selenium and silver concentrations are below a level of concern at all sampling locations.

- ***Ponded Water on the Impoundment.*** At sampling location RF-9, measured concentrations of arsenic and mercury are slightly above respective toxicity values (HQs of 3). Zinc concentrations are also above the toxicity value (HQ of 10). All other COPC concentrations are below levels of concern for amphibians.

- ***Unnamed Drainage flowing into the South Diversion Ditch.*** At sampling location RF-3-2, concentrations of arsenic, copper, and mercury are slightly above respective toxicity values (HQs ranging from 2 to 5). Total zinc concentrations are above the toxicity value with an HQ of 100. Concentrations of all other COPCs are below a level of concern for amphibians.

The range of HQs for amphibians from surface water are summarized below.

Location	As	Cd	Cu	CN	Pb	Hg	Se	Ag	Zn
Silver Creek - upstream	<1 to 5	<1 to 3	2 to 100	<1 to 30	<1 to 400	<1 to 1000	All <1	All <1	800 to 100,000
Silver Creek - downstream	<1 to 8	<1 to 2	<1 to 10	<1 to 20	<1 to 90	<1 to 1000	All <1	All <1	200 to 2,000
Site Ponded Water	3	<1	<1	NC	<1	3	<1	<1	10
South Diversion Ditch	<1 to 200	All <1	<1 to 5	<1 to 8	<1 to 10	<1 to 3	All <1	All <1	90 to 3,000
Unnamed Drainage	3	<1	56	NC	<1	2	<1	<1	100

The HQ values indicate that potential risks for amphibians associated with exposures to arsenic, copper, lead, mercury and zinc in the surface waters of Silver Creek both upstream and downstream of the RFT Site, the South Diversion Ditch, site ponded water and the Unnamed Drainage on the RFT Site. Adverse effects associated with lead, mercury, and zinc (as shown by the size of the ratio and frequency of exceedances) are predicted to be the most severe and frequent.

Figures 7-8a to 7-8e compare data on the distribution (mean and maximum) of typical concentrations of total recoverable concentrations of arsenic, copper, lead, mercury and zinc observed in Silver Creek and in RFT Site surface waters to the range of species toxicity values for amphibians. The toxicity values shown are derived from AWQC Documents (USEPA 1985b-e, 1987, 1996, 2001b) and are presented in Table 7-8. As seen in Figure 7-8a, arsenic concentrations in Silver Creek and in RFT Site waters are all below available toxicity values for amphibians. Copper concentrations (Figure 7-8b), with the exception of station N4, are also all below toxicity levels for available amphibian species. In Figure 7-8c, maximum lead concentrations at stations RF-7-2 in upstream Silver Creek, USC-1 and USC-2 in downstream Silver Creek, and RF-6 and N5 in the south diversion ditch are all above the EC50 for the marrow mouthed toad. Stations N4 and N6 are greater than toxicity values for the leopard frog and marbled salamander, but these concentrations appear to be anomalous in comparison with other measured lead concentrations. Maximum total mercury concentrations (Figure 7-8d) at station RF-7-2 in upstream Silver Creek, station RF-8 in downstream Silver Creek, and RF-4 in the south diversion ditch are above a level of concern for the African clawed frog. Mercury concentrations at stations N4 and N6 are several orders of magnitude above typical concentrations in other surface water, the reason for this discrepancy is not known at this time. Zinc concentrations (Figure 7-8e) at most locations are above the EC50 for the narrow-mouthed toad, but are below a level of concern for the African clawed toad and the marbled salamander with the exception of station RF-7.

7.2.2 Seep Water

HQ values for amphibians are calculated for COPCs in seep water (as estimated from groundwater) and are presented in Table 7-9. The left-hand side of the table presents the maximum total recoverable COPC concentrations from each groundwater monitoring well. If total concentrations are not available, the dissolved concentrations are used to calculate HQs. The corresponding amphibian toxicity benchmark screening values are also presented. Where the HQ values exceed 1E+00, the values are in boldface type.

A summary of the seep water HQ results for each COPC in which an toxicity benchmark exceedance occurred is provided below. A summary of the total HI at each monitoring well and the contribution of each COPC HQ to the total HI is presented in Figure 7-9.

Location	As	Cd	Cu	CN	Pb	Hg	Se	Zn
Seep Water @ Main Embankment	20to90	4 to 10	3 to 400	30 to 50,000	20 to 30	3 to 22	2	70 to 3,000
Background Groundwater	<1	<1	8	20	200	2	<1	100

Inspection of these HQ values shows exceedances of the toxicity values for amphibians to a greater extent for seep waters at the base of the main embankment compared to background waters for almost all COPCs. The highest HQ values are observed for cyanide and zinc, however, seep water concentrations of arsenic, cadmium, copper, lead and mercury also exceed respective amphibian toxicity values indicating potential risk associated with these COPCs.

7.3 Plants

7.3.1 Soil

The basic equation used for calculation of an HQ value for exposure of plants to COPCs in soils is:

$$HQ_{plant} = \frac{Conc_{soil}}{TB_{plant}}$$

where:

$Conc_{soil}$ = Concentration of metal in soil (mg/kg)
TB_{plant} = Phytotoxicity benchmark value (mg/kg) for COPC (Table 6-4)

As discussed previously, HQ values for plants are calculated based on total recoverable COPC concentrations in soil samples from each sampling location. HQ values are calculated based on the low and the high phytotoxicity value (from Table 6-4) for each COPC. These results are presented in Appendix F. If all HQ values based on the low phytotoxicity benchmark are below one, it is concluded that risks for plants associated with direct contact to COPCs in surface soils are not expected. Conversely, if the majority of HQ values based on the high benchmark are substantially higher than one, it is concluded that phytotoxicity is likely.

The HQ results (Appendix F) are summarized graphically in Figure 7-10 by soil type (background, on-impoundment, off-impoundment and tailings). For each COPC, HQs calculated using the low and high phytotoxicity benchmarks (Table 6-4) are presented in the upper and lower panels, respectively. The HQ ranges presented for each general soil type represent the minimum and maximum calculated HQs; the average HQ is also presented. The following table summarizes the HQ values for plants from exposure to COPCs in soil.

Location	Al	Sb	As	Ba	Cd	Cr	Cu	Pb	Hg	Se	Ag	Zn
Background Soils	NA	NA	all <1	all <1	all <1	20	all <1	<1 to 2	all <1	3	<1	2 to 3
Off-Impoundment Soils	NA	NA	<1 to 30	all <1	<1 to 7	20 to 30	all <1	<1 to 100	all <1	3	<1	2 to 30
On-Impoundment Cover Soils	400 to 500	<1 to 2	<1 to 10	all <1	<1 to 2	20 to 40	all <1	<1 to 60	all <1	3	<1	<1 to 20
Site Tailings	40 to 300	9 to 50	<1 to 30	NA	<1 to 10	9 to 30	<1 to 7	2 to 200	all <1	3 to 10	6 to 30	60 to 200

NA = Not Analyzed

- ***Background Soils.*** The concentrations of most COPCs in background soils are below the low toxicity benchmark for plants. These HQs indicate that phytotoxicity is not likely to occur as a result of direct contact with these COPCs in soil. HQ values for chromium, lead, selenium and zinc are all slightly above one, but are lower than HQ values observed for either on-impoundment or off-impoundment soils.

- ***Off-Impoundment Soils.*** The average concentrations of arsenic, chromium, lead, selenium and zinc in off-impoundment soils are above the phytotoxicity benchmarks (HQs ranging from 2 to 100). These HQs indicate that phytotoxicity is likely to occur as a result of direct contact with these COPCs in soil. HQs for barium, copper, mercury and silver are all below one. Cadmium HQs based on maximum concentrations are slightly above one for off-impoundment soils using the low phytotoxicity benchmark.

- ***On-Impoundment Soils.*** Aluminum and chromium HQs for all on-impoundment soils are above a both the low and high phytotoxicity benchmarks (maximum HQ of 500 for aluminum). These HQs indicate that phytotoxicity is likely to occur as a result of direct contact with these COPCs in soil. HQ values for barium, copper, mercury and silver are all below one. HQ values based on the low phytotoxicity benchmark for antimony, arsenic, cadmium are slightly above one, while maximum HQ values for lead and zinc range from 20 to 60.

- ***Tailings.*** HQ values for all COPCs except mercury are above the low phytotoxicity benchmarks. The highest HQs are for lead and zinc (HQs of 200 compared to the low phytotoxicity benchmarks). These HQ values indicate that phytotoxicity is probable if direct contact for plants were to occur with tailings material. The extent of existing soil cover (both depth and extent) as well as the root zone depth of existing vegetation cover is key to understanding if these exposures are possible.

Figure 7-11 presents the contribution of each COPC HQ to the total HI for each general location (background, off-impoundment and off-impoundment). The COPCs which contribute most to the HI are aluminum, lead and zinc. The HQ values depicted in the figure are based on the average soil concentrations of each COPC across available depths at a sampling location.

7.3.2 Seep Water

The basic equation used for calculation of an HQ value for exposure of plants to COPCs in seep water is:

$$HQ_{plant} = \frac{Conc_{soil}}{TB_{plant}}$$

where:

$Conc_{soil}$ = Concentration of metal in soil (ug/L)
TB_{plant} = Phytotoxicity Benchmark Value (ug/L) for COPC (Table 6-5)

HQ values for plants are calculated for COPCs in seep water (as estimated from groundwater) and are presented in Table 7-10. The left-hand side of the table presents the total recoverable and dissolved COPC EPCs from each groundwater monitoring well. The corresponding phytotoxicity screening benchmark for solution exposures for each COPC is also presented. Where the HQ values exceed one, the values are in boldface type. If the HQ exceeds one, then potential risks may occur and there is a need for further evaluation. The calculated HQs for plants from direct contact with seep water are summarized below.

Location	Aluminum	Arsenic	Chromium	Copper	Lead	Manganese	Zinc
Seep Water @ Main Embankment	20 to 300	80 to 300	<1 to 2	<1 to 30	4 to 7	<1 to 4	<1 to 7
Background Groundwater	50	4	all <1	all <1	30	all <1	all <1

Figure 7-12 presents the contribution of each COPC to the total HI at each groundwater monitoring well. The primary contributors to risk at the base of the main embankment are aluminum, arsenic, copper and lead (maximum HQs of 300). Concentrations of these COPCs in upgradient (background) wells are also above the phytoxicity benchmarks. Concentrations of beryllium, cadmium, cobalt, mercury and selenium at all locations are all below a level of concern (HQs < 1). For upgradient (background) groundwater, concentrations of chromium, copper, manganese and zinc are below respective phytotoxicity benchmarks.

These HQ values indicate that risks for terrestrial plants associated with direct contact with aluminum, arsenic, copper and lead in seep water are possible. These HQ calculations are screening level estimates based on estimates of seep water concentrations of each COPC from available groundwater monitoring well data. Conclusions may change in the baseline risk assessment as more information on the extent of contamination of seeps becomes available.

7.4 Soil Fauna

The basic equation used for calculation of an HQ value for exposure of soil fauna to COPCs in soils is:

$$HQ_{soil\ fauna} = \frac{Conc_{soil}}{TB_{soil\ fauna}}$$

where:

$Conc_{soil}$ = Concentration of COPC in soil (mg/kg)
$TB_{soil\ fauna}$ = Toxicity benchmark (mg/kg) for COPC for soil fauna

HQ values are calculated based on the low and high toxicity benchmark for each COPC (Table 6-6). These results are presented in Appendix G for each soil sampling location for each COPC. If all HQ values are below one based on the low toxicity benchmark, it is concluded that risks to soil fauna associated with direct contact to COPCs in surface soils are not expected. Conversely, if the majority of HQ values based on the high benchmark are higher than one, it is concluded that adverse effects to soil fauna toxicity are likely.

The HQ results are summarized graphically in Figure 7-13 by soil type (background, on-impoundment, off-impoundment and tailings). For each COPC, HQs calculated using the low and high toxicity benchmarks (Table 6-6) are presented in the upper and lower panels, respectively. The HQ ranges presented for each general soil type represent the minimum and maximum calculated HQs; the average HQ is also presented. The following table summarizes the HQ values for soil fauna from exposure to COPCs in soil.

Location	Al	As	Ba	Cd	Cr	Cu	Pb	Hg	Se	Ag	Zn
Background Soils	NA	all <1	all <1	all <1	<1 to 60	all <1	all <1	<1 to 2	all <1	all <1	all <1
Off-Impoundment Soils	NA	<1 to 10	all <1	<1 to 20	<1 to 80	<1 to 2	<1 to 40	<1 to 30	all <1	all <1	<1 to 10
On-Impoundment Soils	30 to 40	<1 to 6	all <1	<1 to 4	<1 to 90	<1 to 2	<1 to 20	<1 to 10	all <1	all <1	<1 to 10
Site Tailings	3 to 30	<1 to 20	NA	<1 to 40	<1 to 80	<1 to 20	2 to 80	<1 to 200	<1 to 6	all <1	5 to 90

NA = Not Analyzed

- ***Background Soils.*** The concentrations of most COPCs in background soils are below respective low toxicity benchmarks for soil fauna. These HQs indicate that adverse effects to soil fauna is not likely to occur as a result of direct contact with these COPCs in soil. The HQ values for chromium and mercury are slightly above one, but are lower than HQ values for the Off and On–Impoundment Soils.

- ***Off-Impoundment Soils.*** The average concentrations of arsenic, cadmium, chromium, lead, mercury and zinc in off-impoundment soils are above the low toxicity benchmarks (HQs ranging from 2 to 60). These HQ values indicate that adverse effects to soil fauna is likely to occur as a result of direct contact with these COPCs in soil. HQ values for barium, selenium and silver are all below one. Copper HQs based on maximum concentrations are slightly above one (HQ of 2).

- ***On-Impoundment Soils.*** Aluminum HQ values for on-impoundment soils are above a level of concern (maximum HQ of 40). These HQ values indicate that adverse effects to soil fauna is likely to occur as a result of direct contact with aluminum in soil. HQ values for barium, selenium and silver are all below one. Maximum HQs based on the low toxicity benchmark exceed one for arsenic, cadmium, chromium, copper, lead, mercury and zinc.

- ***Tailings.*** All measured concentrations of aluminum, copper, lead and zinc in tailings are above toxicity benchmarks for soil fauna. Average HQ values for arsenic, cadmium, chromium, mercury and selenium exceed respective low toxicity benchmarks. The highest HQs are observed for mercury (maximum HQ of 200 compared to the low benchmark).

These HQ values indicate that adverse effects to soil fauna is likely if these receptors are exposed to the tailings material under the current soils cover.

Figure 7-14 presents the contribution of each COPC HQ to the total HI for each general soil location (background, off-impoundment, and on-impoundment). The COPCs which contribute most to the HI are aluminum, chromium, lead, mercury and zinc, but other COPCs also contribute to risks. The HQ values in the figure are based on average soil concentrations of each COPC across available depths.

7.5 Wildlife Receptors

7.5.1 Surface Water

Potential risks for wildlife receptors from exposure to COPCs in surface water are characterized by use of the HQ approach. The HQ is defined as the ratio of the dose to the appropriate TRV (Table 6-8):

$$HQ_{sw} = \frac{Dose_{sw}}{TRV_{water}}$$

where:

$Dose_{sw}$ = Average Daily Dose of COPC via ingestion of surface water (mg/kg BW/day)
TRV_{water} = Toxicity reference value for ingestion of water (mg/kg BW/day)

The basic approach used for estimating exposure and risk for wildlife receptors is to estimate the dose and the HQ for each COPC separately, and then to add HQs across all COPCs to derive a hazard index (HI). If the HI is less than or equal to one, no unacceptable risks to the exposed wildlife receptor is assumed. If the value of the HI exceeds one, then potential risks may occur and there is a need for further evaluation.

HI values are presented using both NOAEL and LOAEL TRVs (described in Section 6.5). All HI values are represented to one significant digit. HI values are calculated for each receptor for each exposure area (upstream Silver Creek, downstream Silver Creek, south diversion ditch, ponded water and unnamed drainage) and are summarized in the following text table. The detailed HQ_{sw} values calculated for each COPC are provided in Appendix E for each wildlife receptor.

Receptor	Hazard Indices for Surface Water Ingestion				
	Silver Creek Upstream	Silver Creek Downstream	South Diversion Ditch	Ponded Water	Unnamed Drainages
Cliff Swallow	all < 1	all < 1	all < 1	all < 1	all < 1
Greater-Sage Grouse	**< 1 to 2**	all < 1	all < 1	all < 1	all < 1
Mallard Duck	all < 1	all < 1	all < 1	all < 1	all < 1
Belted Kingfisher	all < 1	all < 1	all < 1	all < 1	all < 1
American Robin	all < 1	all < 1	all < 1	all < 1	all < 1
American Kestrel	all < 1	all < 1	all < 1	all < 1	all < 1
Red Fox	all < 1	all < 1	all < 1	all < 1	all < 1
Masked Shrew	**< 1 to 4**	all < 1	all < 1	all < 1	all < 1
Mink	all < 1	all < 1	all < 1	all < 1	all < 1
Deer Mouse	all < 1	all < 1	all < 1	all < 1	all < 1

As seen, HI values for almost all wildlife receptors are less than one for each exposure area. The HQ values indicate that risks for wildlife related to ingestion of COPCs in surface water are unlikely. The exception is for the greater-sage grouse and the masked shrew at upstream locations on Silver Creek with HIs ranging from <1 to 2 and <1 to 4, respectively. A review of the detailed HQ values presented in Appendix E shows that the majority of the risk observed in the upstream Silver Creek areas is attributable to total concentrations of lead in the surface water.

7.5.2 Sediment

Potential risks for wildlife receptors from exposure to COPCs in sediment are characterized by use of the HQ approach. The HQ is defined as the ratio of the dose associated with ingestion of sediments to the appropriate dietary TRV (Table 6-8):

$$HQ_{sed} = \frac{Dose_{sed}}{TRV_{diet}}$$

where:

$Dose_{sed}$ = Average Daily Dose of COPC via incidental ingestion of sediment (mg/kg BW/day)

TRV_{diet} = Toxicity reference value for dietary exposure (mg/kg BW/day)

HI values are presented in the following text table as a range using both NOAEL and LOAEL TRVs (Table 6-8). HQs are calculated for each COPC for each exposure area (upstream Silver Creek, downstream Silver Creek, south diversion ditch and wetlands area). The detailed HQ_{sed} values calculated for each COPC are provided in Appendix E.

Receptor	Hazard Indices for Sediment Ingestion			
	Silver Creek Upstream	Silver Creek Downstream	South Diversion Ditch	Wetlands Area
Belted Kingfisher	40 to 80	10 to 20	3 to 10	8 to 20
Mallard Duck	40 to 80	10 to 20	3 to 10	8 to 20
Mink	50 to 100	10 to 30	5 to 20	10 to 30

HI values for each receptor exceed one for all exposure areas. Based on relative HI values, the greatest risks are predicted for receptors at upstream locations on Silver Creek. A review of the detailed HQ values presented in Appendix E reveals which COPCs are contributing to the majority of the predicted risk within each exposure area. Figure 7-15 presents the contribution of each COPC to the total HI for sediment ingestion for each wildlife species.

For the belted kingfisher and mallard exposed to COPCs by ingestion in upstream Silver Creek., aluminum, arsenic, cadmium, lead and zinc contribute most to the total HI (Figure 7-15). Most of the total HI is attributable to lead (HQs range from 30 to 70). Aluminum, antimony, arsenic and lead HQs are all greater than one for the mink. Almost all of the total HI for mink is attributable to antimony (HQs range from 20 to 60) and lead (HQs range from 30 to 60).

For downstream Silver Creek, the South Diversion Ditch and the Wetlands Area the HQ values for most COPCs, with the exception of aluminum and lead, are less than one for the belted kingfisher and the mallard. For the mink, aluminum, antimony and lead HQ values are greater than one. All other COPC HQs are less than one. For mink, HQ values for thallium in the wetland area greater than one. In general, the HI values are highest for the wetland area followed by downstream Silver Creek and the South Diversion Ditch.

7.5.3 Seeps

Potential risks for wildlife receptors from exposure to COPCs in seep water are characterized by use of the HQ approach. The HQ is defined as the ratio of the dose to the appropriate TRV (Table 6-8):

$$HQ_{seep} = \frac{Dose_{seep}}{TRV_{diet}}$$

where:

$Dose_{seep}$ = Average Daily Dose of COPC via ingestion of seep water (mg/kg BW/day)
TRV_{diet} = Toxicity reference value for water exposure (mg/kg BW/day)

For the purposes of the SERA, it is conservatively assumed that 100% of the drinking water for each representative species comes from seeps.

HI values are presented in the following text table as a range using both NOAEL and LOAEL TRVs (Table 6-8). HI values are calculated for each representative species for each exposure area (upgradient wells and wells below main embankment). The detailed HQ_{seep} values calculated for each COPC for each representative species are provided in Appendix E. A summary of the results is provided in the following text table.

Receptor	Hazard Indices for Seep Water Ingestion	
	Upgradient Monitoring Wells	Monitoring Wells below Main Embankment
Cliff Swallow	all < 1	all < 1
Greater-Sage Grouse	all < 1	all < 1
Mallard Duck	all < 1	all < 1
Belted Kingfisher	all < 1	all < 1
American Robin	all < 1	all < 1
American Kestrel	all < 1	all < 1
Red Fox	all < 1	all < 1
Masked Shrew	< 1 to 3	all < 1
Mink	all < 1	all < 1
Deer Mouse	all < 1	all < 1

HQs based on the NOAEL and the LOAEL TRV for almost all representative wildlife species are less than one for the ingestion of seep water. The exception is the masked shrew, for which lead HQ values for upgradient groundwater are greater than one (based on the NOAEL TRV) (Figure 7-16). The lead HQ based on the NOAEL TRV for the masked shrew is 3.

7.5.4 Soil

Potential risks for wildlife receptors from exposure to COPCs in soils are characterized by use of the HQ approach. The HQ is defined as the ratio of the dose to the appropriate TRV (Table 6-8):

$$HQ_{soil} = \frac{Dose_{soil}}{TRV_{diet}}$$

where:

$Dose_{soil}$ = Average Daily Dose of COPC via incidental ingestion of soil (mg/kg BW/day)

TRV_{diet} = Toxicity reference value for COPC for dietary exposure (mg/kg BW/day)

The HI values for each representative wildlife species for each exposure area are summarized in the following text table using both NOAEL and LOAEL TRVs (Table 6-8). The detailed HQ_{soil} values calculated for each COPC are provided in Appendix E for each representative wildlife species.

Receptor	Hazard Indices for Soil Ingestion			
	Background Soils	Off-Impoundment Soils	On-Impoundment Soils	Site Tailings
American Robin	2 to 5	10 to 30	20 to 60	70 to 200
American Kestrel	all < 1	< 1 to 2	<1 to 4	5 to 10
Greater-Sage Grouse	all < 1	all < 1	all < 1	<1 to 3
Red Fox	all < 1	< 1 to 2	2 to 10	8 to 20
Masked Shrew	20 to 70	20 to 60	400 to 2,000	3,000 to 8,000
Deer Mice	all < 1	3 to 8	8 to 30	30 to 90

Based on relative HI values, the risks predicted for the masked shrew are the highest observed for any of the representative wildlife species with HI values greater than of one for all exposure areas. The highest risks are predicted for ingestion of tailings with HI values as low as <1 to 3 for the greater sage grouse to a 8,000 for the masked shrew. Risks for exposure to On-impoundment soils is higher than Off-Impoundment soils. The lowest overall risks are predicted for representative wildlife species exposed to soils at areas identified as background.

A review of the detailed HQ values presented in Appendix E reveals which COPCs are contributing to the predicted risk for each exposure area. Figure 7-17 provides a summary of the contribution of COPCs to the HI for each representative wildlife species for each exposure area. These results are discussed in the following paragraphs.

- ***Background Soils.*** HQs and total HIs for the American kestrel, red fox, deer mouse, and greater-sage grouse are all less than one. For the American robin, chromium concentrations are slightly above the selected NOAEL TRV (HQ of 2). Calculated HQs for arsenic, barium, and lead are all greater than one for the masked shrew, with the highest HQ values observed for lead (HQ range from 20 to 50).

- *Off-Impoundment Soils.* Similar to background soils, HQs and total HIs for the American kestrel, the red fox, and the greater-sage grouse are all less than one. HQs for the American robin are greater than one for barium, cadmium, lead, and zinc (maximum HQ of 10). HQs for the masked shrew are greater than 1 for arsenic, barium, cadmium, lead, and zinc. Lead concentrations contributed the most to the total HI. For the deer mouse, only lead HQs (range of 2 to 6) are greater than one. In general, HI values for all representative wildlife species are higher for Off-Impoundment soils compared to background.

- *On-Impoundment Soils.* Total HIs for on-impoundment soils are greater than one for all representative wildlife species except the greater-sage grouse. Aluminum, chromium, and lead HQs contributed most to the total HI. In addition to these COPCs, antimony, arsenic, barium and zinc also contribute significantly to the total HI for the masked shrew. In general, HI values for all representative wildlife species are higher for On-Impoundment soils compared to off-impoundment soils.

- *Tailings.* The total HI values for all representative wildlife species are greater than one. HQ values for lead and antimony contributed the most to the total HI for most species. However, HQs for other COPCs such as aluminum, arsenic, cadmium, chromium, copper, mercury, selenium and zinc also contribute to risks for the American robin and masked shrew. In general, HI values for all representative wildlife species are higher for tailings compared to on-impoundment soils.

HI values greater than one for at least one species within all exposure areas indicate that risks for wildlife related to incidental ingestion of soils is likely. The COPCs which contribute most to excess risks are aluminum, antimony and lead; however, other COPCs are also of concern for the American robin and masked shrew.

7.5.5 Food Chain

Potential risks for wildlife receptors from exposure to COPCs in food chain items are characterized by use of the HQ approach. The HQ is defined as the ratio of the dose to the appropriate TRV (Table 6-8):

$$HQ_{diet} = \frac{Dose_{diet}}{TRV_{diet}}$$

where:

$Dose_{diet}$ = Average Daily Dose of COPC via ingestion of food (mg/kg BW/day)
TRV_{diet} = Toxicity reference value for dietary exposure (mg/kg BW/day)

The five dietary media evaluated in the SERA are ingestion of benthic invertebrates, fish, plants, earthworms, and small mammals. The results for each dietary item are presented in the following subsections.

7.5.5.1 Benthic Invertebrates

The HI values for each representative wildlife species (the mallard duck) consuming benthic invertebrates for each exposure area are presented as a range in the following text table using both NOAEL and LOAEL TRVs (Table 6-8). The detailed HQ_{diet} values calculated for each COPC are provided in Appendix E.

Receptor	Hazard Indices for Benthic Invertebrate Ingestion			
	Silver Creek - upstream	Silver Creek - downstream	Wetlands Area	South Diversion Ditch
Mallard Duck	1,000 to 6,000	400 to 2,000	200 to 2,000	400 to 3,000

The HI values for the mallard are greater than one within all exposure areas with the highest risks predicted for upstream Silver Creek. It is important to note that benthic tissue concentrations are estimated using sediment EPC values and BSAFs (Section 5.3.5.1). Actual tissue concentrations of metals in benthic invertebrates arc cxpected to be lower. For the mallard, HQ values for most COPCs are greater than one based on both NOAEL and LOAEL TRVs. Cadmium, copper, lead and zinc appear contribute to the majority of the predicted risk (Figure 7-18).

7.5.5.2 Fish

The HI values for each representative wildlife species (the belted kingfisher and mink) consuming fish for each exposure area are presented as a range in the following text table using both NOAEL and LOAEL TRVs (Table 6-8). The detailed HQ_{diet} values calculated for each COPC are provided in Appendix E.

Receptor	Hazard Indices for Fish Ingestion			
	Silver Creek - upstream	Silver Creek - downstream	Wetlands Area	South Diversion Ditch
Belted Kingfisher	10,000 to 30,000	4,000 to 8,000	1,000 to 4,000	3,000 to 8,000
Mink	20,000 to 50,000	5,000 to 10,000	2,000 to 6,000	4,000 to 10,000

The HI values for the belted kingfisher and mink are greater than one within all exposure areas with the highest risks predicted for upstream Silver Creek. Aluminum, arsenic, lead and zinc appear to be contributing to the majority of the predicted risk.. Similarly for the mink, HQ values for most COPCs are greater than one, with antimony and lead (maximum HQs of 8,000 and 10,000), contributing most to the total HI (Figure 7-19). It is important to note that fish tissue concentrations are estimated using sediment EPC values and BSAFs (Section 5.3.5.1). Actual tissue concentrations of metals in fish are expected to be lower.

7.5.5.3 Plants

The HI values for each representative wildlife species (deer mouse and Greater-sage grouse) consuming terrestrial plants for each exposure area (background soils, off-impoundment soils, on-impoundment soils are presented as a range in the following text table using both NOAEL and LOAEL TRVs (Table 6-8). The detailed HQ_{diet} values calculated for each COPC are provided in Appendix E.

Receptor	Hazard Indices for Plant Ingestion			
	Background Soils	Off-Impoundment Soils	On-Impoundment Soils	Tailings
Deer Mouse	<1 to 3	3 to 6	2 to 5	20 to 40
Greater-Sage Grouse	all < 1	<1 to 2	all < 1	3 to 9

The HI values for the deer mouse are greater than one within all exposure areas with the highest risks predicted for exposure to plants growing on tailings followed by off-impoundment and on-impoundment soils. Risks to the Greater-sage grouse are predicted to be lower than those for the deer mouse. Within the background and on-impoundment soils exposure areas, all HI values are less than one. Within the off-impoundment and tailings exposure areas the HI values are greater than 1 but no individual HQ value is greater than one.

For both off-impoundment and on-impoundment soils, lead concentrations in plants are the primary risk drivers (Figure 7-20). For tailings, cadmium, lead, selenium and zinc concentrations in plants are the risk drivers. In interpreting the HI values, It is important to note that plant tissue concentrations are estimated using soil EPC values and bioaccumulation factors or models (Section 5.3.5.2). Actual tissue concentrations of metals in plants may be lower or higher.

7.5.5.4 Earthworms

The HI values for each representative wildlife species (American Robin and Masked Shrew) consuming earthworms for each exposure area (background soils, off-impoundment soils, on-impoundment soils are presented as a range in the following text table using both NOAEL and LOAEL TRVs (Table 6-8). The detailed HQ_{diet} values calculated for each COPC are provided in Appendix E.

Receptor	Hazard Indices for Earthworm Ingestion			
	Background Soils	Off-Impoundment Soils	On-Impoundment Soils	Site Tailings
American Robin	30 to 100	100 to 900	70 to 300	500 to 3,000
Masked Shrew	200 to 600	1,000 to 4,000	700 to 2,000	6,000 to 20,000

The HI values for the American robin and the masked shrew are greater than one within all exposure areas with the highest risks predicted for ingestion of earthworms from tailings followed by off-impoundment and on-impoundment soils and then background. Risks predicted for the masked shrew are approximately 10-fold higher than those for the American robin. For both off-impoundment and on-impoundment soils, ingestion of lead concentrations in earthworms is the primary risk drivers (Figure 7-21). For tailings, ingestion of cadmium and lead in earthworm tissues are the primary risk drivers. It is important to note that plant tissue concentrations are estimated using soil EPC values and bioaccumulation factors or models (Section 5.3.5.3). Actual tissue concentrations of metals in earthworm tissues is unknown and may be lower or higher than the estimates used to evaluate risks..

7.5.5.5 Small Mammals

The HI values for each representative wildlife species (American kestrel and red fox) consuming small mammals for each exposure area (background soils, off-impoundment soils, on-impoundment soils are presented as a range in the following text table using both NOAEL and LOAEL TRVs (Table 6-8). The detailed HQ_{diet} values calculated for each COPC are provided in Appendix E.

Receptor	Hazard Indices for Small Mammal Ingestion			
	Background Soils	Off-Impoundment Soils	On-Impoundment Soils	Site Tailings
American Kestrel	4 to 10	10 to 90	6 to 20	20 to 200
Red Fox	<1 to 3	5 to 9	3 to 5	10 to 20

The HI values for the American kestrel and red fox are greater than one within all exposure areas with the highest risks predicted for ingestion of small mammals from the tailings exposure area followed by off-impoundment and on-impoundment soils and then background. Risks predicted for the American kestrel are approximately 10-fold higher than those for the red fox. For both off-impoundment and on-impoundment soils, ingestion of cadmium and lead in small mammals are the primary risk drivers (Figure 7-22). For tailings, ingestion of cadmium, lead and selenium in small mammal tissues are the primary risk drivers. It is important to note that small mammal tissue concentrations are estimated using soil EPC values and bioaccumulation factors or models (Section 5.3.5.4). Actual tissue concentrations of metals in small mammals is unknown and may be lower or higher than the estimates used to evaluate risks.

7.5.6 Wildlife Summary

The results of the SERA indicate a potential for adverse effects to wildlife receptors associated with the ingestion of metals in surface water, sediment, soil, benthic invertebrates, fish, plants, earthworms and small mammals. Based on the evaluation of the HI values in the previous subsections the following is summarized concerning potential risks for wildlife:

- ***Ingestion of Surface Water.*** Risks are predicted only for upstream Silver Creek for the masked shrew and Greater-Sage grouse as a result of ingestion of lead in surface water. All other HI values for wildlife are less than one and below a level of concern.

- ***Ingestion of Seep Water.*** Risks are predicted only for upgradient groundwater for the masked shrew ingesting lead. All other HI values for wildlife are less than one and below a level of concern.

- ***Ingestion of Sediment.*** Total HIs for the mallard, belted kingfisher and mink from the incidental ingestion of sediment are greater than one for all locations in Silver Creek, the south diversion ditch, and the wetlands area. HI values are highest for upstream Silver Creek followed by downstream Silver Creek, the wetlands area, and the south diversion ditch. Lead and aluminum contribute the most to risk for avian receptors while antimony and lead contribute the most to predicted risks for the mink.

- ***Ingestion of Soil.*** Total HIs are greater than one for all avian and mammalian representative species for on-impoundment soils and tailings. HI values are also greater than one for some species for off-impoundment soils. Aluminum and lead contributes the most to predicted risks for on-impoundment soils while lead is the primary contributor to risks for off-impoundment soils. In background soils, arsenic, barium and lead contribute the most to predicted risks for the American robin and the masked shrew. Risks for exposure to On-impoundment soils is higher than Off-Impoundment soils. The lowest overall risks are predicted for representative wildlife species exposed to soils at areas identified as background.

- Ingestion of Benthic Invertebrates. Total HI values for the mallard are greater than one for all exposure areas. The primary contributors to risk are cadmium, lead and zinc. Risks (based on relative HI values) are highest for upstream Silver Creek followed by the south diversion ditch and the wetlands area and downstream Silver Creek.

- ***Ingestion of Fish.*** Total HI values for the belted kingfisher and mink are greater than one for all exposure areas. Aluminum, antimony, lead and selenium contribute most to the predicted risks for the mink. For the belted kingfisher, aluminum, arsenic, cadmium and zinc contribute the most to predicted risks.

- ***Ingestion of Plants.*** HI values are greater than one for both species evaluated (the Greater-sage grouse and the deer mouse) for some exposure areas (off-impoundment soils and tailings). Lead and selenium are the primary contributors to the predicted risks. Risks (based on relative HI values) are highest for tailings followed by off and on-impoundment soils and background.

- ***Ingestion of Earthworms.*** HI values for both representative species are greater than one for all exposure areas. Lead and mercury are the primary contributors to the predicted risks. Risks (based on relative HI values) are highest for tailings followed by off-impoundment soils, on-impoundment soils and background.

- ***Ingestion of Small Mammals.*** Total HI values for both species (the American kestrel and red fox) are greater than one for exposure areas. Cadmium, lead and selenium are the primary contributors to the predicted risks.

7.6 Summary of SERA Results

The primary findings of the SERA for the RFT Site are summarized in Table 7-11. These findings are used to identify the data need to complete a more detailed analyses of ecological risks. These data gaps and recommended data to fill them are discussed further in Section 9.0.

8.0 UNCERTAINTIES

The HQ values presented should not be interpreted as highly precise estimates of actual risk of ecological effects. Quantitative evaluation of ecological risks is limited by uncertainty (lack of knowledge) regarding a number of important data, exposure, toxicity, and risk factors. This lack of knowledge is usually circumvented by making estimates based on whatever limited data are available, or by making assumptions based on professional judgement when no reliable data are available. Because of these assumptions and estimates, the results of the risk calculations are themselves uncertain, and it is important for risk managers and the public to keep this in mind when interpreting the results of a risk assessment.

The USEPA recommends that an ecological risk assessment include a discussion of uncertainties that influence the interpretation of the results (USEPA, 1997). This section summarizes the key sources of uncertainty influencing the results of the SERA. The discussion of uncertainties is organized according to the components of the SERA. A tabular summary is provided in Table 8-1.

8.1 Uncertainties in Problem Formulation

8.1.1 Selection of Receptors

Risks to wildlife are assessed for a small subset of the species likely to be present at the RFT Site. The representative wildlife species selected for quantitative evaluation represent a range of taxonomic groups and life history types. An effort was made to select species representing the full range of possible exposures present in the area. This analyses, however, was completed in the absence of site-specific information concerning wildlife species and habitat present at the RFT site. These species may not, however, represent the full range of sensitivities present. The species selected may be either more or less sensitive to contaminant exposures than typical species located within the area. In particular, the relative sensitivities of reptiles as compared to birds, mammals, or amphibians are unknown. It is assumed that the risks to these organisms are at least qualitatively similar to risks to birds, mammals, and amphibians. Reptile species were not selected, as toxicity data for ingestion exposures to contaminants is limited.

8.1.2 Selection of Exposure Pathways

The exposure pathways selected for evaluation in the SERA are not inclusive of all potential exposure pathways for all ecological receptors. It is necessary to select a subset of possible exposure pathways for two primary reasons: 1) There is not enough information available to evaluate an exposure pathway and 2) it is necessary to limit the effort required when completing the assessment. For the SERA, the pathways selected for analyses are believed to represent those where contaminant exposures are highest.

8.1.3 Exposure Pathways that could not be Evaluated

Certain exposure pathways could not be evaluated in the SERA including:

- Exposures for amphibians to COPCs in soil and dust via inhalation, direct contact or ingestion could not be evaluated due to a lack of dose-response information for these exposure pathways as well as a lack of exposure parameters necessary to estimate COPC doses.

- Exposures for amphibians to COPCs in sediment, surface water, seeps and the aquatic food chain via ingestion could not be evaluated due to a lack of dose-response information for these exposure pathways as well as a lack of exposure parameters necessary to estimate COPC doses.

8.1.4 Selection of Ecological Contaminants of Potential Concern (COPCs)

The methodology used to select COPCs in the SERA may result in a number of uncertainties. These uncertainties are outlined below.

- Risk evaluation is only completed for those contaminants that have been identified as COPCs through the screening process. Not evaluating contaminants that are not identified as COPCs, but for which data are available may result in a slight underestimate of risk.

- Contaminants that are not detected, but for which the detection limit exceeds a level of concern are identified as a source of uncertainty. USEPA (1989) suggests eliminating those contaminants that have not been detected in any samples of a particular medium, although the detection limits exceed levels of ecological concern. It is assumed that these contaminants would only have a negligible effect on risk levels and would not likely result in a significant underestimate of risk.

- Contaminants with a detection frequency less than five percent are identified as a source of uncertainty. It is assumed that the infrequent presence of these contaminants would have only a negligible effect on risk levels and would not likely result in a significant underestimate of risk.

- Although a reference (background) comparison screening step for inorganics is identified in the COPC selection process, this reference comparison is not effectively used in the selection process as the sample sizes for all reference data sets are too small (sample size less than five) or are not representative of background.

8.2 Uncertainties in Exposure Assessment

8.2.1 Environmental Concentrations

In the exposure assessment, the desired input parameter is the true mean concentration of a contaminant within a medium, averaged over the area where exposure occurs. For the RFT Site, environmental data were not obtained in a truly random fashion and are likely to be biased toward areas of maximum contamination. In addition, the available data sets for the SERA are currently incomplete, which provides a limited means for deriving reliable exposure estimates.

The techniques used for data sampling and analysis, and methods used for selecting contaminants for evaluation in the risk assessment may result in a number of uncertainties. These uncertainties are itemized below.

- Analyzed samples may not represent the actual levels of contaminants at the RFT Site. This may result in either an over- or underestimate of risk.

- Systematic or random errors in the contaminant analyses may yield erroneous data. These types of errors may result in a slight over- or underestimate of risk.

- The UCL95 or maximum concentrations are used to represent levels of exposure for terrestrial wildlife. Use of these upper bound concentrations provides a conservative estimate of average RFT Site concentrations; actual exposures may, however, be lower or higher.

8.2.2 Lack of Data on Extent of Contamination in Seeps

Analytical data for the seep located at the base on the main embankment are not available. Aquatic and terrestrial receptors may be exposed to contamination in the seeps via direct contact or ingestion. Groundwater data from several monitoring wells near the seep were used to evaluate possible risks associated with the seeps. Use of the groundwater data may result in either an under- or overestimation of risks.

8.2.3 Limited Data on the Extent of Contamination in the Wetlands

Surface water and sediment data for the wetlands area located west of the main embankment are limited. Previous reports indicate that the wetland sediments are tailings (ESE, 1993). Aquatic and terrestrial wildlife may be exposed to contamination in sediment and surface water in the wetlands by direct ingestion or the ingestion of food. The SERA analyses is limited to 4 sediment samples from the wetland. Use of these limited data may result in either an under- or overestimation of risks.

8.2.4 Limited Analyses of Soil Samples

Soil samples (either on impoundment or off impoundment) were analyzed for all metals in only 20% of the samples collected. All samples were analyzed for both arsenic and lead. This limits the data set for soils for ecological risk analyses and may result in either an under or overestimation of risks as lead and arsenic are not the only COPCs of concern for ecological receptors to soil contamination and do not represent the COPCs associated with the highest risk.

8.2.5 Lack of Data on Extent of Contamination in Biological Tissues

The most direct way to assess dietary exposures for ecological receptors is to measure tissue burdens of COPCs. This measurement eliminates uncertainties associated with estimating the uptake and transfer of contamination from soils, surface water, sediments, and seeps into either the aquatic or terrestrial food chain. Currently, data are not available on tissue concentrations of COPCs in any biological tissues at the

RFT Site. The lack of data may result in either an under- or overestimation of risks. Collection of data on tissue burdens of COPCs would reduce the uncertainties. Collection of tissue samples concurrently with soil and/or sediment samples would provide correlation of tissue burdens with environmental concentrations.

8.2.6 Wildlife Exposure Factors

Ingestion-related exposure assumptions for wildlife are based on literature-derived information concerning average body sizes, diet compositions, consumption rates, and metabolic rates. Much of this information is derived from laboratory-reared animals and may not be representative of feral organisms. Moreover, the actual diet composition of an organism will vary daily and seasonally. These uncertainties could either under- or overestimation the actual exposures of wildlife to COPCs in water, sediment, soil and diet.

8.2.7 Estimation of Doses for Terrestrial Wildlife

Estimates of wildlife exposure due to incidental sediment ingestion conservatively assume that 100% of the metals present are biologically available (100% will be ingested and absorbed in the gut). This assumption likely overestimates contaminant doses to wildlife, as absorption efficiencies for most metals are less than 100%.

It is also assumed in the calculation of contaminant doses for wildlife that contaminants present in environmental media have the same bioavailability as contaminants in laboratory test media. This assumption is conservative because laboratory testing purposely includes dosing regimes (method of administration and contaminant form) to insure a uniform and maximum uptake of contaminants.

8.3 Uncertainties in Effects Assessment

8.3.1 General Use of Toxicity Screening Benchmarks

The literature-derived data used to identify toxicity benchmarks contain uncertainties related to the application of generic data to site-specific conditions. The toxicity benchmarks identified for the SERA are based on data from a wide range of sites and conditions, many of which may be quite different from the conditions at the RFT Site. These literature-derived values are expected to be less accurate than site-specific data, but the magnitude and direction of any errors introduced by their use are unknown.

There are often important site-specific factors that may tend to modify (often decrease) the toxicity of metals in surface water, sediments and soil. In general, these site-specific factors are referred to as "bioavailability" factors. For example, metals in surface water may be bound to soluble organic materials that reduce the tendency for the metal to bind to respiratory structures of fish or benthic organisms. Similarly, the presence of organic matter in soil, along with other substances, may have a significant influence on actual toxicity. One of the best ways for investigating the importance of such factors is to perform toxicity tests using site-specific media, either by in-situ assays or laboratory bioassays. The results of site-specific toxicity studies can significantly increase the accuracy of the ERA process.

8.3.2 General Use of Sediment Toxicity Benchmarks

A potential limitation to the use of sediment screening benchmarks is that not all of the metals in the bulk sediment may be available for dissolution into the pore water. Studies by a number of researchers have found that the tendency of certain metals in sediment to dissolve into the pore water is determined in large part by the amount of sulfide present in the sediment (Hansen et al., 1996; Ankley, 1996; Ankley et al., 1996). This is because divalent cations of heavy metals such as cadmium, copper, lead, zinc, and nickel form highly insoluble complexes with sulfides. Thus, if the sediment contains sufficient sulfide to complex the metals, then dissolution into pore water and resultant toxicity to benthic organisms is not expected (Hansen et al., 1996; Ankley, 1996; Ankley et al., 1996).

Based on these considerations, one method for evaluation of site-specific effects and risks for benthic invertebrates to metals in sediments is to measure the amount of acid-extractable cadmium, copper, lead, zinc, nickel, and mercury (these are referred to as Simultaneously Extractable Metals, or SEM). The SEM is compared to the simultaneously measured level of Acid Volatile Sulfide (AVS). If the measured level of SEM (mmol/g) is the same or less than AVS (mmol/g), then it is expected that the metals in sediment are not contaminantly available to partition to pore water. Thus, toxicity to benthic invertebrates is not of concern. If the concentration of SEM exceeds the concentration of AVS, then there is a possibility of metal release to pore water and possible toxicity. An exceedance of AVS by SEM is not proof that toxicity will occur, especially if the exceedance is fairly small (e.g., less than approximately 5 mmol/g) (Hansen et al. 1996). This is due to the observation that other materials in sediment (e.g., organic carbon) may also bind metals (Mahony et al., 1996; Hansen et al., 1996).

Another direct method for measuring exposure and assessing risks for sediment-dwelling benthic invertebrates is to measure the concentration of metals in the sediment pore water and to compare those measurements to appropriate screening benchmarks or to complete toxicity testing in the laboratory exposing test organisms to site whole phase sediment samples.

8.3.3 Absence of Toxicity Benchmarks

Toxicity screening benchmarks were not available for all COPCs. A summary of these unavailable benchmarks is provided below. The lack of these benchmarks may result in the under-estimation of potential risks.

Absence of Toxicity Screening Benchmarks	
Type of Benchmark	**COPC**
Ambient Water Quality Criteria for Aquatic Receptors for Direct Contact with Water	Antimony, barium, beryllium, boron, calcium, cobalt, lithium, magnesium, manganese, molybdenum, potassium, sodium, strontium, thallium or vanadium Chronic Criteria Only: iron Acute Criteria Only: silver
Toxicity Benchmarks for Amphibians for Direct Contact Exposures with COPCs in surface water or seep water.	Boron, thallium, vanadium, cobalt, chromium, manganese,
Toxicity Benchmarks for Benthic Invertebrates for Direct Contact Exposures to COPCs in Sediment	Barium, beryllium, cobalt, selenium, thallium, vanadium
Toxicity Benchmarks for Soil Fauna for Direct Contact Exposures to COPCs in Soil	Antimony

8.3.4 Absence of Wildlife TRVs

Avian toxicity data for antimony and silver were not available in either of the secondary review sources (Sample et al., 1996 and Engineering Field Activity West, 1998). Quantitative assessments of risks to avian species related to exposure to antimony and thallium were not performed. This uncertainty results in an underestimation of risks.

8.3.5 Derivation of Wildlife TRVs

Toxicity information for many contaminants is often limited. Consequently, there are varying degrees of uncertainty associated with the wildlife toxicity reference values. These uncertainties may result in an over- or underestimate of risk. Sources of uncertainty associated with toxicity values are listed below.

- Uncertainty in toxicity factors arises from the lack of knowledge on the potential interactive effects of different contaminants. Most TRV values are derived from studies of the adverse effects of a single contaminant. However, exposures to ecological receptors usually involve multiple contaminants, raising the possibility that synergistic or antagonistic interactions might occur. This sort of interaction is of particular importance with regard to metals, since it is known that the absorption and toxicity of some metals interact in complex ways. However, data are not adequate to permit any quantitative adjustment in toxicity values or risk calculations based on inter-contaminant interactions. This uncertainty may result in over- or underestimates of risk.

- Using dose-response information from effects observed at high doses to predict the adverse effects associated with lower doses may result in a slight to moderate overestimate of risk.

- Using dose-response information from short-term exposures to predict the effects of long-term exposures, and vice-versa may result in a slight to moderate over- or underestimate of risk.

The TRVs, in general, are conservative. The NOAEL and LOAEL TRVs are estimated based on: 1) a toxicity value selected from the available scientific literature; and 2) a series of uncertainty factors that account for extrapolation from the laboratory study result (for the toxicity value) to a TRV for a specific representative wildlife species. The process results in an inherently conservative TRV, as the toxicity values selected are the lowest from the reported range.

8.4 Risk Characterization

A baseline ecological risk assessment for the RFT Site should focus on the receptors, exposure pathways and COPCs identified in the SERA as potentially contributing to risks or for which risks could not be estimated. The full characterization should consider both predictive methods (HQs) and site-specific information (biological community measurements including toxicity studies) in a weight-of-evidence evaluation. The data gaps are further described in Section 9.

9.0 DATA GAPS AND RECOMMENDATIONS

The following sections describe the data gaps present in the SERA that need to be filled to complete a quantification of ecological risks. The data gaps are discussed according to potential ecological receptor and exposure medium. The potential exposure media include surface water, sediments, soils and diet. The results of the SERA are summarized in Table 7-11 and are used to discuss data gaps which are described in Table 9-1. The data gaps and recommendations are segregated into analytical data, toxicological data and biological data requirements. Each is discussed with regard to exposure areas on the RFT site including silver creek, the wetland and embankment area, the diversion ditches, on-impoundment soils and off-impoundment soils.

9.1 Silver Creek

Upon examination of the HQ values calculated and presented as Figure 7-1, 7-2 and 7-5 it evident that the surface waters and sediments of Silver Creek are more contaminated with metals upstream of the RFT site compared to downstream. The SERA results show there are risks for aquatic life and terrestrial wildlife for Silver Creek both upstream and downstream of the RFT Site. However, it is concluded based on the SERA information that the RFT Site is not contributing to increased risks in Silver Creek based on both the available surface water and sediment data. Based on this conclusion, further sampling and risk evaluation of Silver Creek in relation to the RFT Site not recommended.

This recommendation, however, is based on the assumption that the risks occurring in Silver Creek will be addressed as part of the risk evaluation of the upstream sources and that any decisions concerning actions in Silver Creek will consider possible influences of future contaminant transport from RFT into Silver Creek. For example, if the metals present in sediments and/or surface water are reduced in Silver Creek (as a result of clean up activities) then the possible discharges from the RFT Site could decontaminate the surface water or sediments and/or become the dominant influence on metal loading.

This recommendation is also exclusive of the reported flood plains tailings pile located immediately west of the tailings impoundment and covering about 6 acres along the banks of Silver Creek (USEPA, 1991). This source is reported to be located on the western side of Silver Creek about 300 feet upstream of the confluence of Silver Creek with the wetland area and extends from there for about 2500 feet upstream. The USEPA and the State of Utah both observed tailings entering Silver Creek from the flood plain tailings pile during site visits for the HRS Scoring (USEPA, 1991). According to analyses performed in 1985 and 1989, the flood plain tailings pile contains arsenic, cadmium, copper, lead, mercury, silver, and zinc (USEPA, 1991). The HRS data is excluded from the SERA as not representative of current conditions on the RFT site in the main impoundment area. Outside of the main impoundment area there could be as many as five samples from this tailings area but the locations are not known (Figure 3-1). It is also not clear from current site boundary information if this area is now part of the RFT Site. This is identified as a data gap for the baseline ERA. If the floodplain tailings are part of the RFT Site then this area needs to be further investigated and recommendations will be provided at a later date.

9.2 Wetland Area and Embankment

9.2.1 Analytical Data

There is currently no data available on the extent of contamination in the surface waters of the wetland area or the seeps at the base of the embankment. The SERA used groundwater data to screen for possible risks associated with the discharge of contaminated groundwater to the wetland. These results for aquatic receptors (Figure 7-6) and amphibians (Figure 7-9) show possible risks for both of these receptor groups associated with exposure to metals in seep water. More accurate risk estimates could be calculated if analytical data from the seeps and the surface water of the wetland were available.

It is recommended that sampling of surface water and seep water be completed to provide data for risk analyses for aquatic receptors, wildlife and amphibians. The samples should be analyzed for the Target Analyte List (TAL) metals. In addition to the analytical measurements, general water quality data should be collected including dissolved oxygen, conductivity, alkalinity, dissolved solids, total suspended solids and pH. All of these factors influence either the potential toxicity of metals or habitat quality.

Additional sediment samples are also recommended from the wetland area. The SERA results are based on evaluation of only 4 sediment samples. Additional sediment samples with concurrent measurements of metals in sediment pore water should be also be collected. The sediment samples would be analyzed for TAL metals and the pore water samples for both total and dissolved TAL metals. The pore water analytical results will be used to evaluate the potential bioavailability and toxicity of the metals in sediment.

9.2.2 Biological Data

There is currently no information available describing the type of wetlands present or habitat. This information is critical to understanding what ecological receptors (aquatic and terrestrial) are using this area of the RFT site and what the possible exposure pathways may be. The type of wetland habitat available determines use by wildlife. It is recommended that the wetland area be surveyed to identify the type of plants present as well as any signs of wildlife use. This survey would be qualitative in nature with the purpose of describing the type of habitat present. This information would then be used to identify the possible species of wildlife present.

Sampling of macroinvertebrates from the wetland area is recommended to identify what species are present within the area and may be exposed to contamination in seeps, surface water and sediments. This information will also be used to possible site-specific toxicity testing.

9.2.3 Toxicological Data

The SERA results predict that surface water, seep water and sediments in the wetland area are likely to be toxic to aquatic receptors. However, site-specific toxicity of the COPCs in these environmental media is not known and could be potentially very different from that predicted. It is recommended that site-specific toxicity testing of environmental media be considered after collection of the basic habitat information along with the goals of the overall RI/FS program. These results will reduce uncertainties in the conservative screening calculations used the SERA and can be used to identify the need for and focus remediation efforts to reduce risks., The toxicity of the seeps to aquatic life could be directly testing using standard surface water toxicity tests with either the daphnid (*Ceriodaphnia dubia*) and/or the fathead minnow (*Pimephales promelas*). The results of these tests would provide direct evidence concerning the toxicity of the seep water and its contribution to surface water toxicity in the wetland. This of course assumes that the wetland habitat present does support aquatic receptors (including amphibians) for at least a portion of the year.

The analytical data presented in this SERA shows substantial contamination of sediments within the wetland area. ESE (1993) concluded that the sediments in the wetland were equal to tailings material. The mean PEC Quotients calculated for wetland sediments (Section 7.2.2.2) indicate that probability of observing toxicity is 100% for 3 of the 4 samples and 88% for a fourth sample. Based on the results of the habitat survey and the use of the area by aquatic and terrestrial wildlife, it may be useful to measure site-specific sediment toxicity using EPA standard whole phase sediment toxicity test protocols with either chironomids and/or the amphipod (*Hyallela azteca*). Toxicity testing of sediment pore water samples is also an option using the standard surface water toxicity testing discussed with regard to toxicity of seeps.

9.2.4 Biological Tissue Data

Risks are predicted for wildlife receptors in the SERA for ingestion of aquatic food items (fish and benthic invertebrates) based on the estimated concentrations of COPCs in these items using existing BSAF models are a conservative assumption. Ingestion of fish and benthic invertebrates contributes the most of the HI values for each representative species (Figure 9-1) compared to incidental ingestion of sediments. The models and assumptions (i.e., ratio of 1:1 between sediment to fish tissue) are conservative and likely overestimate the site-specific uptake of metals and risk.

The bioavailability and uptake of metals from surface water, sediments, and food in the aquatic environment for metals is driven by many site-specific factors. For some contaminants, in particular, mercury and selenium, uptake is not driven by concentrations in sediment and/or water but instead is driven by site-specific microbial activity that controls conversion of the metals from inorganic to organic forms. based on site-specific factors that are difficult to impossible to predict. To reduce the uncertainties in these risk estimates, it is recommended that measurements of metals concentrations be made in aquatic food items available from the wetland for terrestrial wildlife species. Decisions concerning the selection of plant and aquatic organism species for collection and analyses will be dependant on the outcome of the habitat survey.

9.3 South Diversion Ditch

[Table 9-1 complete but not text]

9.3.1 Analytical Data

Current sampling of the sediments of the South Diversion ditch is adequate for establishing extent of contamination. It may however be necessary to collect further samples for analyses concurrently with any toxicity testing, benthic invertebrate sampling, or biological tissue sampling as discussed in the next sections. Samples should be analyzed for TAL metals. Also, sampling and analyses of TAL metals in sediment pore water may be useful in understanding the bioavailability and potential toxicity of metals measured in bulk sediment samples.

9.3.2 Biological Data

Information on the type of habitat provided by the South Diversion ditch and thus its potential use by wildlife and aquatic receptors is unknown. Collection of qualitative data on vegetative cover of the South Diversion Ditch area is recommended to evaluate possible used by wildlife and aquatic receptors. Also recommended is a qualitative sampling of the diversion ditch (concurrently with sediment and sediment pore water samples) to identify the presence or absence of macroinvertebrates and the possible use of the ditch by fish species. Species will be identified to lowest taxonomic level possible.

9.3.3 Toxicological Data

[Table complete but not text]

9.3.4 Biological Tissue Data

[Table complete but not text]

9.4 On and Off-Impoundment Soils

[Table complete but not text]

10.0 REFERENCES

Adriano DC. 1986. *Trace Elements in the Terrestrial Environment.* Springer-Verlag, New York. 553 pp.

Ankley, G. T., D. M. Di Toro, and D. J. Hansen and WD. J. Berry. 1996. Technical Basis and Proposal for Deriving Sediment Quality Criteria for Metals. *Environ. Toxicol. Chem.* 15:2056-2066.

Ankley, G.T. 1996. Evaluation of Metal/Acid-Volatile Sulfide Relationships in the Prediction of Metal Bioaccumulation by Benthic Macroinvertebrates. *Environ. Toxicol. Chem.* 15:2138-2146.

Agency for Toxic Substances and Disease Registry (ATSDR). 1994. *Preliminary Public Health Assessment Addendum for Richardson Flat Tailings.* Agency for Toxic Substances and Disease Registry, Remedial Programs Branch, Division of Health Assessment and Consultation.

Bechtel Jacobs Company, LLC. 1998. *Biota Sediment Accumulation Factors for Invertebrates: Review and Recommendations for the Oak Ridge Reservation.* Prepared for the US Department of Energy, Office of Environmental Management. August 1998. ORNL Publication BJC/OR-112.

Bromfield, C.S. and M.D Crittenden, Jr. 1971. *Geologic Map of the Park City East Quadrangle, Utah.* US Geological Survey Map GQ-852. Scale 1:24,000. [as cited in RMC, 2000 Watershed SAP]

Brooks, L.E., Mason, J.E., and D.D. Susong. 1998. *Hydrology and Snowmelt Simulation of the Snyderville Basin, Park City, and Adjacent Areas, Summit County, Utah.* US Geological Survey, Water-Resources Investigation Report. [as cited in USEPA, 2001 Watershed Sampling Report]

Canadian Council of Ministers of the Environment (CCME). 1997. *Recommended Canadian Soil Quality Guidelines.* March 1997.

CH2MHill. 1987a. *Assessment of the Toxicity of Arsenic, Cadmium, Lead and Zinc in Soil, Plants and Livestock in the Helena Valley of Montana for East Helena Site (ASARCO)*, East Helena, Montana. EPA Work Assignment No. 68-8L30.0. May.

CH2MHill. 1987b. *Assessment of the Toxicity of Copper, Mercury, Selenium, Silver and Thallium in Soil, Plants and Livestock in the Helena Valley of Montana for East Helena Site (ASARCO)*, East Helena, Montana. EPA Work Assignment No. 68-8L30.0. November.

Dames & Moore. 1980. *Report of Tailings Pond Investigation, Near Park City, Utah*: Consultant's report prepared for Noranda Mining, Inc., November, 1980.

Dames & Moore. 1974. *Report of Embankment and Dike Design Requirements, Proposed Tailings Pond Development, Near Park City, Utah.* Consultant's report prepared for Park City Ventures Corporation, December, 1974.

Ecology and Environment (E&E). 1993. *Richardson Flat Tailings, Summit County, Utah.* Final Report. TDD #T08-9204-015. February 19, 1993.

Ecology and Environment (E&E). 1991. Record of communication with Utah Division of Wildlife Resources. Ecology & Environment. [as cited in ATSDR, 1994 Public Health Assessment]

Ecology and Environment (E&E). 1987. Revised Analytical Results Report of Air Sampling at Richardson Flat Tailings Park City, Utah. TDD R8-8608-05. September 9, 1987. [as cited in RMC, 2000 Watershed SAP]

Edwards, C.A. 1992. Testing the Effects of Chemicals on Earthworms: The Advantages and Limitations of Field Tests. In: *Ecotoxicology of Earthworms*. P.W. Greig-Smith et al. (eds) Intercept Ltd. pp. 75-84.

Efroymson, R.A., M.E. Will, G.W. Suter II, and A.C. Wooten. 1997a. *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants: 1997 Revision.* Prepared for the U.S. Department of Energy, Office of Environmental Management by Lockheed Martin Energy Systems, Inc. managing the Oak Ridge National Laboratory (ORNL). ORNL publication ES/ER/TM-85/R3, November.

Efroymson, R.A., M.E. Will and G.W. Suter II. 1997b. *Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision.* Prepared for the U.S. Department of Energy, Office of Environmental Management by Lockheed Martin Energy Systems, Inc. managing the Oak Ridge National Laboratory (ORNL). ORNL publication. ES/ER/TM-126/R2, November.

Eijsackers, H. 1994. Ecotoxicology of Soil Organisms: Seeking the Way in a Pitch-Dark Labyrinth. In: *Ecotoxicology of Soil Organisms*. M.H. Donker, H. Eijsackers and F. Heimbach. Eds. CRC Press.

Engineering Field Activity, West. 1998. *Development of Toxicity Reference Values for Conducting Ecological Risk Assessments at Naval Facilities in California, Interim Final.* EFA West, Naval Facilities Engineering Command, United States Navy. San Bruno, California.

Hansen, D.J., W. J. Berry, J. D. Mahoney, W. S. Boothman, D. M. Di Toro, D. L. Robson, G. T. Ankley, D. Ma, Q. Yan, and C. E. Pesch. 1996. Predicting the Toxicity of Metal-Contaminated Field

Sediments Using Interstitial Concentration of Metal and Acid-Volatile Sulfide Normalizations. *Environ. Toxicol. Chem.* 15:2080-2094.

Ingersoll, C. G., P.S. Haverland, E. L. Bruson, T. J. Canfield, F. J. Dwyer, C. E. Henke, N. E. Kemble, D. R. Mount, and R. G. Fox. 1996. Calculation and Evaluation of Sediment Effect Concentrations for the Amphipod *Hyallela azteca* and the Midge *Chironomus riparius*. *Great Lakes Res*. 23:602-623.

IRIS. 1998. Retrieval from USEPA's Integrated Risk Information System (IRIS). (http://www.epa.gov/ngispgm3/iris/index.html)

Kabata-Pendias, A. and H. Pendias. 1992. *Trace Elements in Soils and Plants*, 2nd Edition, CRC Press, 365 pgs.

Long, E. R., and L. G. Morgan. 1991. *The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program.* National Oceanic and Atmospheric Administration. NOAA Technical Memorandum NOS OMA 52. Seattle, WA. August, 1991.

Long, E.R. and L.G. Morgan. 1990. *The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program.* NOAA Technical Memorandum NOS OMA 52. Seattle, Washington. March.

Long E.R., D.D. MacDonald , J.C. Cubbage, and C.G. Ingersoll. 1998. Predicting the Toxicity of Sediment-Associated Trace Metals with Simultaneously Extracted Trace Metal: Acid-Volatile Sulfide Concentrations and Dry Weight-Normalized Concentrations: A Critical Comparison. *Environ. Toxicol. Chem.* 17:972-974.

Long, E. R., D.D. MacDonald, S. L. Smith and F. D. Calder. 1995. Incidence of Adverse Biological Effects within Ranges of Chemical Concentrations in Marine and Estuarine Sediments. *Environmental Management* 19(1):81-97.

MacDonald, DD, CG Ingersoll, and TA Berger. 2000. Development and Evaluation of Consensus-Based Sediment Quality Guidelines for Freshwater Ecosystems. Arch Environ Contam Toxicol 39:20-31.

MacDonald, D. D., R. S. Carr, F. D., E. R. Long and C. G. Ingersoll CG. 1996. Development and evaluation of sediment quality guidelines for Florida coastal waters. *Ecotoxicology* 5:253-278.

MacDonald Environmental Sciences, Ltd. 1994. *Approach to the Assessment of Sediment Quality in Florida Coastal Waters. Volumes I to IV.* Prepared for the Florida Department of Environmental Protection. November.

Mahoney, J. D., D. M. Di Toro, A. M. Gonzales, M. Curto, M. Dilg, L. D. Rosa and L. A. Sparrow. 1996. Partitioning of Metals to Sediment Organic Carbon. *Environ. Toxicol. Chem.* 15:2187-2197.

Mason, J.L. 1989. Hydrology of the Prospector Square Area, Summit County, Utah: US Geological Survey, Water-Resources Investigation Report 88-4156, 75 pp. [as cited in RMC, 2000 Watershed SAP]

Pedigo, N. G., W. J. George, and M. B. Anderson MB. 1988. Effects of Acute and Chronic Exposure to Cobalt on Male Reproduction in Mice. *Reproductive Toxicology*. 2:45-53.

Persaud, D., R. Jaagumuagi, and A. Hayton. 1993. *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario.* Water Resources Branch, Ontario Ministry of the Environment and Energy. August, 1993.

Prothro M. 1993. Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria. Memorandum to Water Management Division Directors and Environmental Services Directors, Regions I-X. October 1, 1993.

Resource Management Consultants (RMC). 2001a. *Sampling and Analysis Plan for the Remedial Investigation Richardson Flat.* February 20, 2000.

Resource Management Consultants (RMC). 2001b. *INTERIM DRAFT - Remedial Investigation/ Feasibility Study.* December 5, 2001.

Resource Management Consultants (RMC). 2000a. *Focused Remedial Investigation/Feasibility Study Work Plan.* May 25, 2000.

Resource Management Consultants (RMC). 2000b. *Sampling and Analysis Plan for the Upper Silver Creek Watershed.* May 1, 2000.

RIVM. 1997. *Maximum Permissible Concentrations and Negligible Concentrations for Metals, Taking Background Concentrations into Account.* Crommentuijn et al. Rijksinstituut Voor Volksgezondheid en Milieu. National Institute of Public Health and the Environment. Report No 601501 001. Bilthoven, the Netherlands.

Sample, B.E., J.J. Beauchamp, R.A. Efroymson, G.W. Suter, II, and T.L. Ashwood. 1998a. *Development and Validation of Bioaccumulation Models for Earthworms.* Oak Ridge National Laboratory. ES/ER/TM-220. February, 1998.

Sample, B.E., J.J. Beauchamp, R.A. Efroymson, and G.W. Suter, II. 1998b. *Development and Validation of Bioaccumulation Models for Small Mammals.* Oak Ridge National Laboratory. ES/ER/TM-219.

Sample, B. E., D. M. Opresko, and G. W. Suter II. 1996. *Toxicological Benchmarks for Wildlife: 1996 Revision.* Prepared for the U.S. Department of Energy, Office of Environmental Management by Lockheed Martin Energy Systems, Inc. managing the Oak Ridge National Laboratory (ORNL). ORNL publication. ES/ER/TM-86/R3, June.

Skorupa, J. P. 1998. Selenium Poisoning of Fish and Wildlife in Nature: Lessons from Twelve Real-World Examples. In: *Environmental Chemistry of Selenium*. William Frankenberger, Jr. and Richard A. Engberg. Eds. Markel Dekker, Inc.

United States Environmental Protection Agency (USEPA). 2001a. *Data Interpretation Report - Upper Silver Creek Watershed Surface Water/Stream Sediment Monitoring 2000.* Prepared by the USEPA incorporation with the Upper Silver Creek Watershed Stakeholders Group. Final - May 4, 2001.

United States Environmental Protection Agency (USEPA). 2001b. *2001 Update of Ambient Water Quality Criteria for Cadmium.* EPA-822-R-01-001. April 2001.

United States Environmental Protection Agency (USEPA). 1998. *Guidelines for Ecological Risk Assessment. Risk Assessment Forum.* Office of Research and Development, Washington, D.C. EPA/630/R-95/002F. April 1998.

United States Environmental Protection Agency (USEPA). 1997. *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. Interim Final.* U.S. Environmental protection Agency, Environmental Response Team (Edison, NJ). June 5, 1997.

United States Environmental Protection Agency (USEPA). 1996. *1995 Updates: Water Quality Criteria Documents for the Protection of Aquatic Life in Ambient Water.* U.S. Environmental Protection Agency, Office of Water. September, 1996. EPA-820-B-96-001.

United States Environmental Protection Agency (USEPA). 1995. *Water Quality Standards; Establishment of Numeric Criteria for Priority Toxic Pollutants; States' Compliance - Revision of Metals Criteria.* 40 CRF 131 (60) pp22228-22237. May 4, 1995.

United States Environmental Protection Agency (USEPA). 1993a. *Wildlife Exposure Factors Handbook. Volume I.* U.S. Environmental Protection Agency, Office of Research and Development. EPA/600/R-93/187a. December 1993.

United States Environmental Protection Agency (USEPA). 1993b. *Wildlife Exposure Factors Handbook. Volume II.* U.S. Environmental Protection Agency, Office of Research and Development. EPA/600/R-93/187a. December 1993.

United States Environmental Protection Agency (USEPA). 1993c. *Memorandum to ATSDR concerning EPA review of Initial Release Public Health Assessment.* March 9, 1993. [as cited in ATSDR, 1994 Public Health Assessment]

United States Environmental Protection Agency (USEPA). 1992. *Framework for Ecological Risk Assessment. Risk Assessment Forum.* Office of Research and Development, Washington, D.C. EPA/630/R-92/001. February 1992.

United States Environmental Protection Agency (USEPA). 1991. *National Priorities List for Richardson Flat Tailings, Summit County, Utah.* OERR Hazardous Site Evaluation Division, Washington, D.C.

United States Environmental Protection Agency (USEPA). 1989. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response. *Risk Assessment Guidance for Superfund. Volume I. Human Health Evaluation Manual (Part A).* USEPA Document EPA.

United States Environmental Protection Agency (USEPA). 1987. *Ambient Water Quality Criteria for Zinc - 1987.* U.S. Environmental Protection Agency, Office of Water. EPA 440/5-87-003. February, 1987.

United States Environmental Protection Agency (USEPA). 1985a. *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses.* U.S. Environmental Protection Agency. Office of Research and Development. January, 1985. NTIS Document Number PB85-227049.

United States Environmental Protection Agency (USEPA). 1985b. *Ambient Water Quality Criteria for Arsenic - 1984.* U.S. Environmental Protection Agency, Office of Water. EPA 440/5-84-033. January, 1985.

United States Environmental Protection Agency (USEPA). 1985c. *Ambient Water Quality Criteria for Cadmium - 1984.* U.S. Environmental Protection Agency, Office of Water. EPA 440/5-84-032. January, 1985.

United States Environmental Protection Agency (USEPA). 1985d. *Ambient Water Quality Criteria for Copper - 1984.* U.S. Environmental Protection Agency, Office of Water. EPA 440/5-84-031. January, 1985.

United States Environmental Protection Agency (USEPA). 1985e. *Ambient Water Quality Criteria for Lead - 1984.* U.S. Environmental Protection Agency, Office of Water. EPA 440/5-84-027. January, 1985.

Warren L, Tessler A, Hare L. 1998. Modeling Cadmium Accumulation by Benthic Invertebrates In Situ: The Relative Contribution of Sediment and Overlying Water Compartments to Organism Cadmium Concentrations. *Limnol. & Oceanog.* 43: (in press).

Witschi, H. and J.A. Last. 1987. *Handbook of Toxicology.* T.J. Haley and W. O. Berndt, (Eds). Washington D.C. Hemisphere Publishing Corporation, pages 112-156.

Weston Engineering, Inc. (Weston). 1999. *Preliminary Hydrogeologic Review of Richardson Flat Tailings Site, Summit County, Utah.* Prepared for United Park City Mines.

TARGET SHEET
EPA REGION VIII
SUPERFUND DOCUMENT MANAGEMENT SYSTEM

DOCUMENT NUMBER: 2008741

SITE NAME: RICHARDSON FLAT TAILINGS

DOCUMENT DATE: 02/01/2002

DOCUMENT NOT SCANNED

Due to one of the following reasons:

☐ PHOTOGRAPHS

☐ 3-DIMENSIONAL

☐ OVERSIZED

☐ AUDIO/VISUAL

☐ PERMANENTLY BOUND DOCUMENTS

☐ POOR LEGIBILITY

☐ OTHER

☐ NOT AVAILABLE

☑ TYPES OF DOCUMENTS NOT TO BE SCANNED
(Data Packages, Data Validation, Sampling Data, CBI, Chain of Custody)

DOCUMENT DESCRIPTION:

FIGURES, TABLES & APPENDICES (See Table of Contents)